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■

OF INTEREST TO MANAGERS

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This issue reviews the zooplankton and fish abundance trend information that sparked the Pelagic Organism Decline (POD) work of 2005. It also incorporates revised mysid and zooplankton indices, which provide a different – less dire – view of lower trophic level responses in 2004 than preliminary analyses presented at a February 2005 Estuary Ecology Team meeting. In particular, the sharp down-turn in 2004 of the key copepod *Pseudodiaptomus forbesi* was not apparent in the revised and recalculated indices. See Lee Mecum's article in the Status and Trends section for more details.

Quarterly Highlights starts with an announcement from Brad Tom, Kate Le and Chris Enright that the Dayflow database now contains water year 2004 data. They also note that the Dayflow computational scheme was modified to estimate X2 during the negative Delta outflow period associated with the Jones Track levee breach (see also Kate Le's second article on Water Year 2005).

Next, three updates from the high profile Collection, Handling, Transport and Release (CHTR) program report successes developing study designs and methodologies. This program is investigating means of improving fish survival from collection fish salvage facilities through release back into the Delta.

In the final Highlight, Janet Thompson informs us of a name change for the infamous Asian clam, *Potamocorbula amurensis*; the clam that changed the Estuary's pelagic food web after its introduction circa 1987. She suggests we now refer to it as *Corbula* (*Potamocorbula*) *amurensis* to avoid confusion and because of likely continued taxonomic revision. Readers will find both *Potamocorbula* and *Corbula* used in subsequent articles.

The Status and Trends section starts with a retrospective of 2004 water-year flows and exports from Kate Le that begins to "set the environmental stage", describing conditions that influenced invertebrate and fish abundance at the freshwater end of the Estuary. Complementary information on 2004 ocean conditions, provided in the Fishes of San Francisco Estuary article by Hieb, Bryant, Dege, Greiner, Souza and Slater, completes the environmental stage information. The generally lower outflows in 2004 as compared to 2003 did not improve upper Estuary habitat, whereas ocean conditions were generally favorable to marine spawners. Details are provided in the next three articles.

Amidst the decline of pelagic organisms in the upper Estuary, *Cancer* crabs residing in the mid to lower estuary did fine in 2004; mitten crabs did not. In her article on crabs of San Francisco Estuary, Kathy Hieb discusses the positive influences a generally cool ocean regime and weak Davidson current have had on *Cancer magister* .

The article Fishes of the San Francisco Estuary contains the 2004 pelagic fish indices that were the third in a series of low annual indices, reinforced concerns about their declines and led to the intensive Pelagic Organism Decline POD investigative work conducted in 2005. These annual indices provide relative measures of current year-class size for comparison to previous ones, and together with indices from other trophic levels have been used to monitor estuarine conditions. Hieb and others show that the Estuary's fishes don't all respond the same: some lower estuary pelagic and bottom dwelling (demersal) fishes have been doing well in the past few years, the marine pelagic northern anchovy has not; a few upper estuary demersal fishes have increased modestly in the past couple years, the pelagic fishes have remained low or declined.

The recent good returns of adult Chinook salmon reported by Erin Chappell reflect primarily historical river conditions (3-4 years previous) and more recent favorable ocean conditions, moderated by ocean commercial and in-river recreational catches. In particular, winter and spring-run stocks have rebounded from low levels, and might be responding to improved in-river temperature conditions and removal of barriers, respectively.

Kate Le, in the final Status and Trends article, provides Water-Year 2005 outflow and export information through March, the former a basis for predicting year-class strength of flow related species. In 2005, San Joaquin River flows improved substantially, but Sacramento River flows declined compared to 2004.

Improved knowledge of the migratory patterns of juvenile hatchery steelhead might help reduce the impacts of south Delta export facilities on hatchery stocks and pro-

vide insights to benefit wild Central Valley steelhead stocks (state and federally threatened) as well. In the first examination if its kind, Steve Foss reviews salvage patterns and transit times of coded-wire-tagged (CWT) juvenile hatchery steelhead. Although transit times were related to the distance between release and salvage locations, but not to outflow during migration, Steve concluded that the low and variable proportions of steelhead coded-wire tagged year to year and hatchery to hatchery limited the knowledge gained from their recovery. He suggests a constant fraction of hatchery fish be CWT.

The three remaining contributed papers delve further into the use and interpretation of long-term monitoring information. In their investigation into developing sub-regions for the Delta, Jassby, Mueller-Solger and Vayssieres continue their critical review of the Environmental Monitoring Program, using the programs water quality data as a means for delimiting sub-regions to estimate phytoplankton biomass and production trends. Though the process didn't produce stable sub-regions, it provided valuable insight into the data and the program, and valuable conclusions drawn that will guide subsequent investigation.

Wim Kimmerer and Bill Bennett have been at the forefront of evolving thought and investigation into the mechanisms underlying the X2 - abundance relationships of a number of estuarine organisms. The current article presents their logic and approach to discriminate among possible mechanisms for several species. The current use of X2 as a water/habitat quality standard and the water costs involved provide the impetus to understand underlying mechanisms in effort to refine the current standards, at hopefully lower water costs.

A persistent criticism of fish abundance indices derived from trawl data (like those presented by Hieb and others) has been the lack of a measure of precision. Concern has been that sampling variability may be so high that abundance indices cannot follow population trends well. Wim Kimmerer and Matt Nobriga directly address this criticism using a data re-sampling method known as bootstrap. Their results show that trawl indices are sufficiently precise to follow trends and in many cases distinguish year to year variation. They also recommend additional refinements to index calculation.

IEP QUARTERLY HIGHLIGHTS

January-March 2005

DAYFLOW 2004 Update

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The DAYFLOW database has been extended to include water year 2004. The output and documentation files are now available at http://www.iep.ca.gov/dayflow/ . This year, the DAYFLOW computational scheme was modified to calculate X2 during the Jones Tract levee breach.

DAYFLOW is a computer program developed in 1978 as an accounting tool for calculating historical Delta outflow and other internal daily average Delta flows. DAYFLOW output is used extensively in studies by State and federal agencies, universities, and consultants.

X2 Estimation and Methods

Flow estimates into Jones Tract as a result of the June 3rd levee breach were provided by DWR's Emergency Response and Beneficial Reuse Unit. The flow estimates for the period of June 3, 2004 through June 5, 2004 were large enough to result in negative values of Delta Outflow (QOUT). The autoregressive lag equation used to calculate X2 cannot compute an X2 value because the log of a negative number is not defined. An analysis of EC data at several locations was performed to estimate X2 as the location of 2 ppt TDS[1,2] for this time period. It was determined that X2 was between Sacramento River at Pittsburg and San Joaquin River at Antioch.

Details of the X2 estimation and methods are available at http://www.iepca.gov/dayflow/output/index.html

Click on the button under "Comments" for WY 2004.

References

1. Dayflow web site: http://www.iep.ca.gov/dayflow/

2. DWR WSIHIST Document: http://cdec.water.ca.gov/cgi-progs/iodir/wsihist/

3. DWR Salinity Unit Conversion Document: http://www.iep.ca.gov/suisun/facts/salin/index.html

Assessment of Fish Predation Occurring in the Collection, Handling, Transport, and Release Phase(CHTR) of the State Water Project's John E. Skinner Delta Fish Protective Facility Fish Salvage Operation

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Preliminary data from 2003 were analyzed to determine the sample size for stomach content comparisons. Results were obtained with the assistance from Mark Bowen of the Bureau of Reclamation. Since the sampling design is of a two sample type, a sample size test for a simple t-test was utilized. This sample size test indicated that >107 samples collected were needed to determine significant differences. An additional 15% would have to be added to the sample size if the data for 2005 is nonparametric. These sample size results are preliminary and are currently being reviewed by DFG Statistician, Phil Law.

The QAPP and SOP protocols have been completed and were tested during dry run experiments in March and April. Only minor additions were made to protocols. Striped bass constituted >90% of predators during dry runs. White catfish and black crappie also occurred in the samples. The occurrence of listed species such as spring run Chinook salmon may restrict implementation of this study and modification to the project's take limits are being investigated. Personnel were trained on sampling methods, fish care, hoist training, and truck driving. The modifications of the CHTR Holding Facility and 2500 gallon tanker truck were completed in March and staff has begun work scheduled from April through July 2005.

Acute Mortality and Injury of Delta Smelt Associated with Collection, Handling, Transport, and Release (CHTR) at the State Water Project and Central Valley Project Fish Salvage Facilities

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This project evaluates the effects of the CHTR phase of the fish salvage process at the South Delta export facilities using injected groups of cultured adult delta smelt. I oversaw the final modifications to the water supply system and fish release pool at the fish testing building at the Skinner Fish Facility in Byron, CA. DFG staff was trained on proper procedures for handling delta smelt and for injecting and recovering test fish during experiments. Quality assurance documents were refined and finalized during this period as well as coordination of testing schedules with the other CHTR study elements. Preliminary testing using adult delta smelt was initiated in March 2005 followed by formal testing at the start of April. Formal testing using adult delta smelt will continue through the middle of April at the Skinner Fish Facility.

DFG and USBR staff continued to evaluate the fluorochrome calcein for mass marking adult delta smelt during the first quarter of 2005. Preliminary results have shown 100% mark retention and 90%-100% survival of marked fish 35 days after marking. The DFG and USBR staff also began evaluating the fluorochrome alizarin red S as a potential marking compound for mass marking adult delta smelt. Although preliminary results indicate that alizarin red S marks delta smelt, further refinement of marking methods and mark detection will continue. Evaluations using juvenile delta smelt will begin in mid-April 2005 using calcein and alizarin red S.

Development of Diagnostic Indicators to Predict Acute or Chronic Adverse Effects to Salvaged Delta Smelt

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This study investigates clinical methods of stress assessment on delta smelt (Hypomesus transpacificus) subjected to the Collection, Handling, Transport, and Release (CHTR) phases of the salvage process at the Skinner Delta Fish Protective Facility.

Preliminary statistical results of the cortisol (the primary stress hormone) analysis from the 2004 pilot work data have been run by DFG statistician Phil Law. The 2004 work has also been used to check the statistical power of sample sizes used.

The 2005 experiments are currently underway and began in late March with wild adult delta smelt. Twenty CHTR stress experiments are planned for the remainder of the 2005 adult season using both wild and cultured adults. Thirty plasma samples from the first four experiments have already been collected and will be analyzed as soon as contracts with the UCD Clinical Endocrinology lab are in place.

Potamocorbula amurensis Is, For Now, *Corbula amurensis*

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When Carlton et al. (1990) published the first paper on *Potamocorbula amurensis* in San Francisco Bay, they cautioned that the Corbulid genera and species were in need of extensive revision and that the species name was in particular likely to change, pending revision of the family. In 2002, Coan published a description of the Eastern Pacific Corbulids and concluded that the San Francisco Bay species was of the genus *Corbula* and that the species name was still unclear (Coan 2002). Based on these find-

ings, the new Lights Manual will list *P. amurensis* as *Corbula amurensis* (James Carlton, personal communication, see "Notes"). To alleviate confusion by local readers, consider referencing this bivalve as *Corbula* (*Potamocorbula*) *amurensis*.

References

Carlton, J.T., J.K. Thompson, L.E. Schemel, and F.H. Nichols. 1990. Remarkable invasion of San Francisco Bay (California, USA) by the Asian clam *Potamocorbula amurensis*. I. Introduction and dispersal. Marine Ecology Progress Series 66:81-94

Coan, E. V. 2002. The Eastern Pacific recent species of the Corbulidae (Bivalvia). Malacologia 44(1): 47-105.

Notes

Carlton, J. T. (Professor, Director of Maritime Studies Program of Williams College). 8 April 2005. E-mail communication.

STATUS AND TRENDS

Water Year 2004 Status and Trends

Kate Le (DWR), kle@water.ca.gov

River Flows and Net Delta Outflow Index

The hydrologic conditions for water year 2004 started off normally. During the period of October to December of 2003, Sacramento River flow, San Joaquin River flow, and NDOI were below 500 cubic meters per second as shown in Figure 1. Thereafter, the amount and frequency of precipitation increased resulting in an increase of Sacramento River flow and NDOI flow to peak at 1,670 and 1,970 cubic meters per second, respectively, as shown in Figure 1. From early January 2004 to mid-February 2004, both Sacramento River flow and NDOI fluctuated between 300 and 2,000 cubic meters per second as a result of decreased and less frequent amount of precipitation events. The largest peak of Sacramento River flow and NDOI occurred in late February and early March of 2004

with an amount of about 2,000 and 5,100 cubic meters per second, respectively. Thereafter, both flows decreased and continue to do so for the remainder of the water year. From June 3 to 5, 2004, an unusual flood event at Jones Tract resulted NDOI to become negative for three days as shown in Figure 1.

San Joaquin River flow as shown in Figure 1 was stable throughout the 2004 water year, and ranged between 30 and 150 cubic meters per second. San Joaquin flow during October 2004 to early January 2005 compared to previous year period was similar. However, from mid-January 2005 to mid-March 2005, San Joaquin flow was about 200 cubic meters per second higher than pervious year for the same period.

Monthly average comparison of flow levels of Sacramento River, San Joaquin River, and NDOI during October 2004 through mid-March 2005 (i.e. 04-05 year) to that of October 2003 through mid-March 2004 (i.e. 03-04) are as follows.

- October: Sacramento River, San Joaquin River, and NDOI flows were higher in 04-05 than 03-04 year. Sacramento flow was about 28 cubic meters per second (1,000 cfs) higher, San Joaquin flow was about 57 cubic meters per second (2,000 cfs) higher, and NDOI was about 96 cubic meters per second (3,400 cfs) higher.
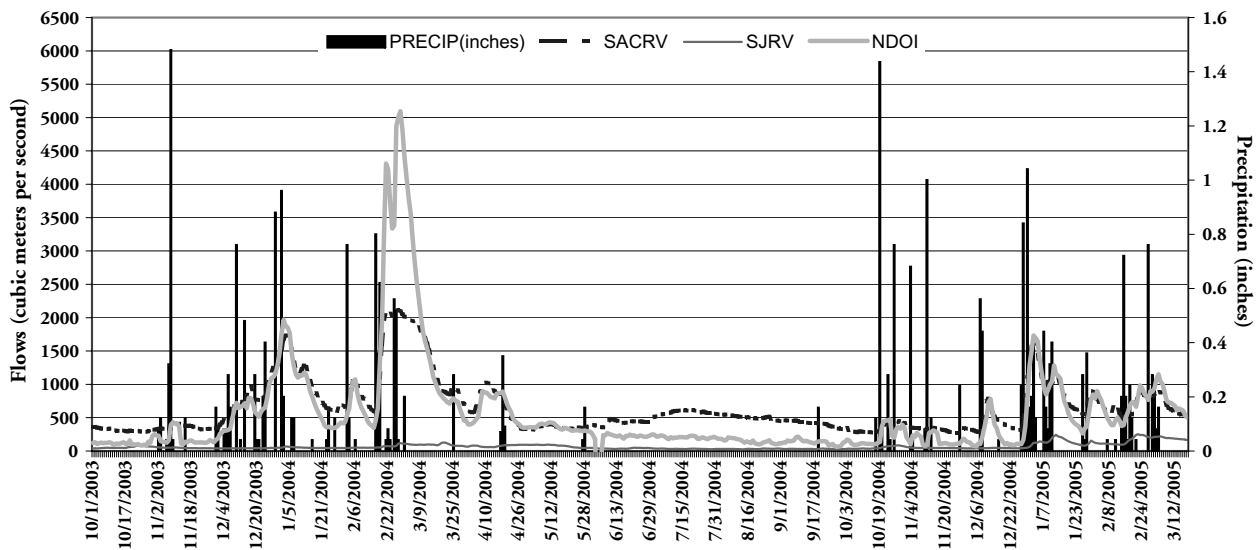

Figure 1   Sacramento River, San Joaquin River, Net Delta Outflow, and Precipitation, October 2003 through mid-March 2005.

- November: Both river flows and NDOI were slightly lower in 04-05 than 03-04 year.

- December: San Joaquin River flow for 04-05 year was very similar to 03-04 year. However, Sacramento River flow and NDOI flows were lower in 04-05 than 03-04 year. Sacramento River and NDOI flows were, 10 times and 13 times lower, respectively, in 04-05 than 03-04 year.

- January: Both river flows and NDOI were lower in 04-05 than 03-04 year; Sacramento River and San Joaquin river flows were about 113 cubic meters per second (4,000 cfs) lower than previous year, whereas NDOI was about 57 cubic meters per second (2,000 cfs) lower in 04-05 than 03-04 year.

- February: San Joaquin River flow was about 88 cubic meters per second (3,100 cfs) higher in 04-05 than 03-04 year, whereas, Sacramento River flow and NDOI were about 566 cubic meters per second (20,000 cfs) and 577 cubic meters per second (20,380 cfs) lower in 04-05 than 03-04 year.

- March to mid-March: San Joaquin River flow was 2 times higher in 04-05 year than 03-04 year, but Sac River and NDOI flows were, 2.4 times and 2.8 times lower, respectively, in 04-05 than 03-04 year.

Exports

During water year 2004, export actions at both SWP and CVP as shown in Figure 2 were operated to meet out-flow standard in October 2003, and water quality standard from November 2003 to January 2004. From February to mid-April of 2004, decreased pumping at CVP was to meet the EI ratio standard. Export actions at both SWP and CVP decreased in mid-April to mid-May in support of VAMP, and thereafter remained low until June for EWA fish protection. At times during the VAMP period, CVP pumping was zero for maintenance reasons. In June 2004, SWP and CVP pumping decreased to meet water quality concerns, whereas in July only the SWP pumping was low to meet the X2 standard.

Monthly average comparison of export levels at SWP and CVP during October 2004 through mid-March 2005 (i.e. 04-05 year) to that of October 2003 through mid-
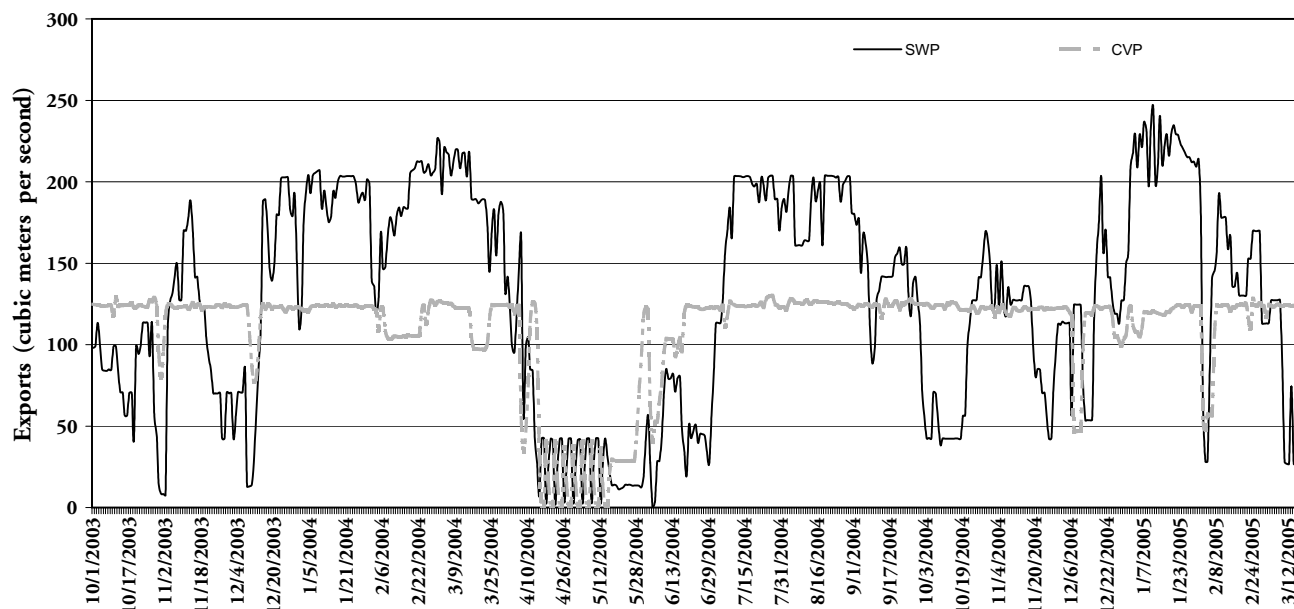
March 2004 (i.e. 03-04 year) are as follow and are shown in Figure 3:

- October, November, and December: Export actions at both SWP and CVP were similar 03-04 and 04-05 years.

- January: SWP pumping was higher (i.e. 28 cubic meters per second or 1,000 cfs more) in 04-05 year than 03-04 year, whereas CVP pumping was lower (i.e. 2.8 cubic meters per second or 100 cfs less) in 04-05 than 03-04 year.

- February: CVP monthly average pumping was about the same in 03-04 and 04-05 years, however, the decrease pumping at CVP in February 2005 was due to Delta smelt action, whereas in February 2004 it was due to EI ratio. SWP pumping was lower (i.e. 42 cubic meters per second or 1,500 cfs less) in 04-05 than 03-04 year.

- March to mid-March: SWP monthly average pumping in 03-04 was 2.3 times higher than 04-05 due to no fishery restriction or curtailment imposed in 03-04 year. CVP pumping was similar in 03-04 and 04-05.

Figure 2 State Water Project and Central Valley Project Pumping, October 2003 through mid-March 2005

Precipitation

Monthly average precipitation of November and December months were similar to each other when comparing 2003-2004 and 2004-2005 years as shown in Figure 3. For October, January, and mid-March months, the monthly average were lower in 2003-2004 than 2004-2005 year. February was the only month where the monthly average in 2003-2004 was higher than 2004-2005.

Percent of Inflow Diverted

Figure 4 is a plot of the 3-day and 14-day percent inflow diverted. During water year 2004, all percent diverted were met for the year. From October 2003 through January 2004, the standard was 65% with the 3-day running average as the controller. From February to June of 2004, the standard was 35% with the 14-day running average as the controller.

WY 2004 Annual Totals

Water year 2004 (October 2003 through September 2004) annual totals are calculated and shown in Figure 5 for the following parameters:

Sacramento River Flow = 17.10 MAF

San Joaquin River Flow = 1.37 MAF
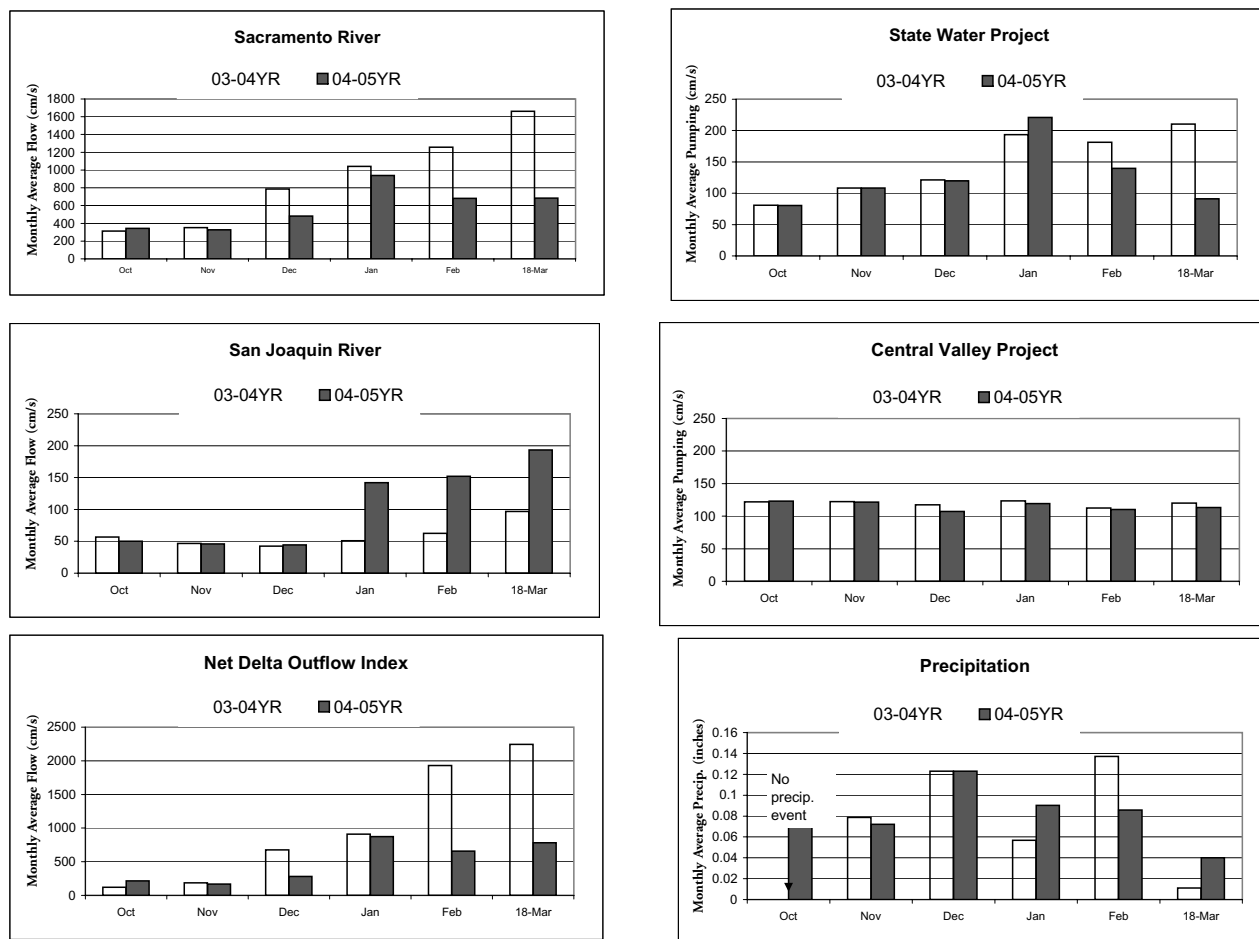
Net Delta Outflow Index = 14.90 MAF

State Water Project = 3.24 MAF

Central Valley Project = 2.72 MAF

SWP + CVP = 5.95 MAF

Figure 3 Comparison of Monthly Average River Flows, Exports, and Precipitation Between 2003-2004 and 2004-2005
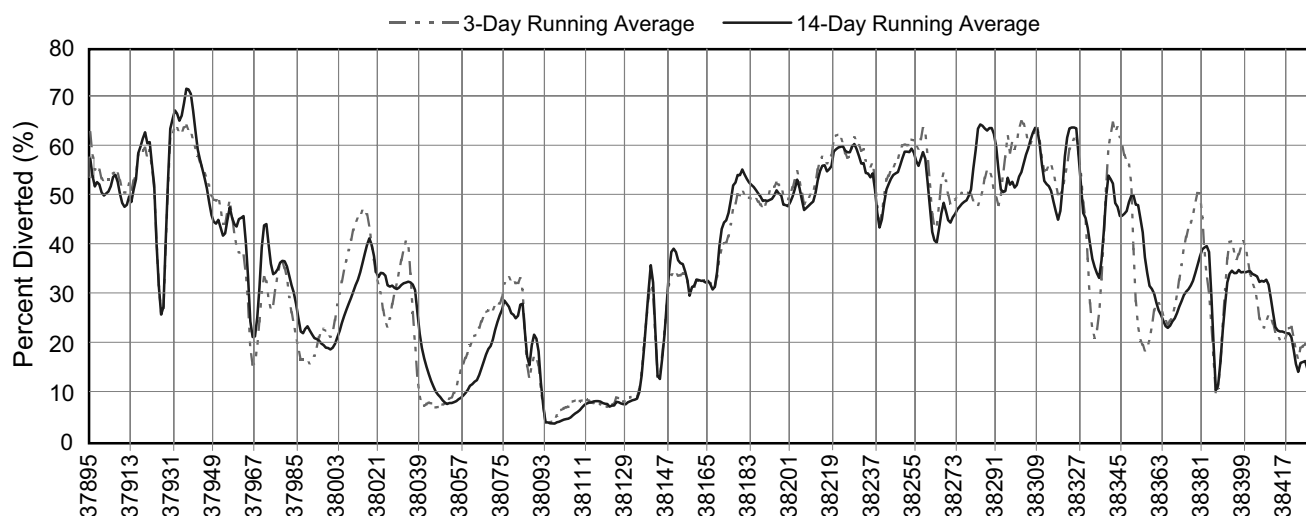

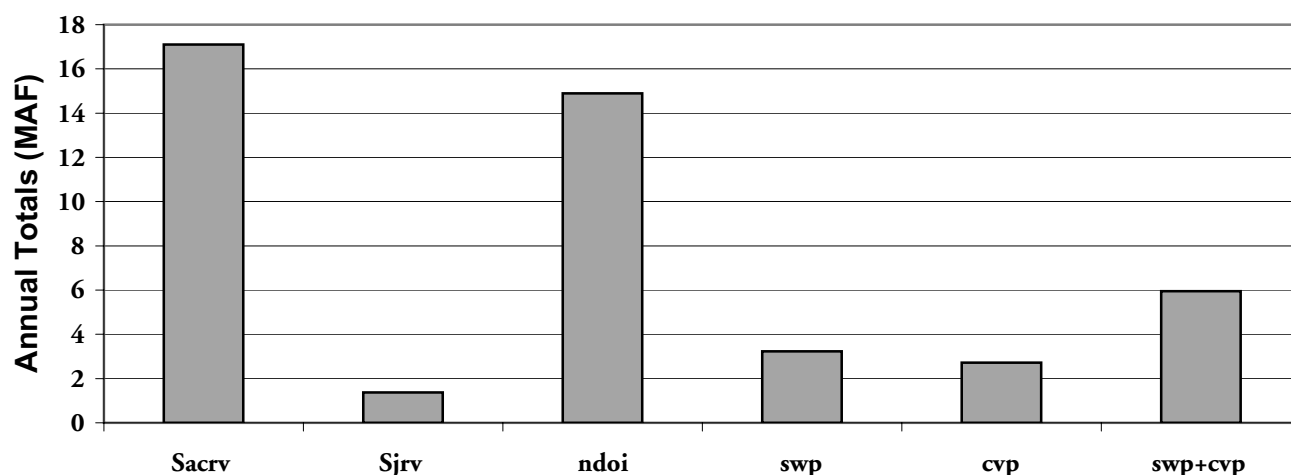**Figure 4 Percent Inflow Diverted, October 2003 through mid-March 2005**

Figure 5 Annual Totals for Water Year 2004 (October 2003 through September 2004)

Zooplankton Monitoring

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The term zooplankton includes animals of varying lengths – rotifers, which average 0.1-0.2 mm, cladocerans, 0.6-2.0 mm, and copepods 1.0-1.2 mm total length for adults. Mysid shrimp are considered to be macro-zooplankton and range from 2 to 18 mm in total length. The Zooplankton Study has estimated the annual abundance, by season, of various zooplankton species or genera in order to assess the size of the fish food resource since 1972. All of the native zooplankters of the upper estuary have decreased in abundance since they were first monitored. In addition, many copepods and several mysids have been introduced to the estuary.

Zooplankton samples were simultaneously taken with phytoplankton samples monthly at the discrete Environmental Monitoring Program sampling sites. In addition to the fixed sites, 2 "floating" entrapment zone stations were sampled monthly where the bottom electrical conductance was 2 and 6 mS/cm.

Three different gears were deployed at each site: a macro-zooplankton net for mysids, a modified Clarke-Bumpus (CB) net for meso-zooplankton, and a pump for micro-zooplankton. Abundance indices were calculated for each gear as the mean number per cubic meter by season and year for all core stations (stations sampled since the inception of the study), plus the 2 floating stations. Seasons were defined as: 1. Spring, March through May, 2. Summer, June through August, and 3. Fall, September through November.

This report incorporates corrections made to the mysid/zooplankton database during summer 2005 and uses single gear-type indices, which is different than past reports. In the past, the indices were derived as the sum of the CB net and pump indices for all non-mysid taxa. Here the indices are reported for only the gear that collects the taxon most efficiently, which is the macro-zooplankton net for all mysids, the CB net for adult copepods and cladocerans and the pump for rotifers. The one exception is *Limnoithona tetraspina*, which is reported for both the CB and pump because both gears catch this species efficiently.

Abundance changes from 2003 to 2004 were mixed, as nearly as many taxa decreased as increased in abundance. Since its introduction in late 1993, the cyclopoid copepod *Limnoithona tetraspina* has been numerically the most abundant copepod in the upper estuary (Figure 1). It is most abundant in Suisun Marsh, Suisun Bay and the lower Sacramento River. Spring pump abundance peaked in 1998 and has been declining ever since, with the spring

2004 index less than half the 2003 value. However, pump abundance increased through summer and by fall, it was about the same as in fall 2003. Summer pump abundance was relatively stable until 2000, but increased in 2001, and has been high ever since; summer 2004 abundance was the third highest summer index recorded. Fall pump abundance has been stable and high since 1999 with only a slight drop in 2003 and 2004. The CB net indices increased from 2003 in all 3 seasons. The spring and fall 2004 indices were the highest indices since 1994 while the summer 2004 index was second only to the summer 1995 value.

Eurytemora affinis is an introduced calanoid copepod that was has been in the estuary before monitoring started. Spring abundance has decreased overall since 1972, but has increased slightly since 2001 (Figure 2). Summer abundance has been quite variable since the introduction of *Pseudodiaptomus forbesi*. The 2004 index was one of the lowest indices since 1989. Fall abundance has also been decreasing since 1972. This trend became particularly steep in the late 1980s, possibly due to competition for food and predation by the introduced clam, *Corbula amurensis*. Since 1995, fall abundance has gradually increased and the fall 2004 index was higher than either the 2002 or 2003 indices.

The spring 2004 abundance of *Pseudodiaptomus forbesi*, an introduced calanoid copepod first recorded in summer 1988, increased considerably from 2003 (Figure 2). In spite of this increase, there appears to be a strong, though variable, downtrend since peak abundance in 1992. For summer and fall, the downward trends have been more gradual than spring, with 2004 indices almost identical to 2002 and 2003. The causes of this decline are unknown.

Several species of the native calanoid copepod genus *Acartia* enter Suisun Bay and the delta from the lower bays as salinity increases seasonally. Because of their brackish water distribution, *Arcartia* are strongly influenced by outflow, such that high outflows can cause the population to move seaward of the sampling area, artificially reducing the abundance index. Prior to 1995, spring abundance was somewhat stable with the exception of major declines coincident with high spring flows (Figure 3). High spring flows in 1995 and 1996 also suppressed abundance. Beginning in 1997, spring abundance increased annually until it reached the second highest recorded level in 2003, but it again decreased in 2004.

Summer and fall abundance has been less variable than spring, but both gradually declined prior to 1995 and have been increasing since. The summer 2004 index was somewhat greater than the 2003 index, while the fall 2004 index was slightly lower.

Acartiella sinensis is an introduced brackish water calanoid copepod, first recorded in spring 1994, that is most abundant in Suisun Bay. Its spring abundance was highly variable prior to 2002. After 3 years of low abundance from 1999 to 2001, spring abundance increased in 2002 and has been stable since (Figure 3). In 2004, summer abundance increased the highest level in 4 years. Fall abundance remained the same as for the 3 previous years.

Several native freshwater species of the calanoid copepod family Diaptomidae occur in the estuary. Their abundance has declined in all seasons since the late 1970s and early 1980s, especially in summer and fall (Figure 4). After a dip in 2003, spring 2004 abundance returned to about the 1999 to 2002 levels. Virtually none were caught in summer 2004 and the abundance index was one of the lowest on record. In contrast, the fall 2004 index was the highest recorded in 15+ years and was very similar to the early decline indices of the mid-1980s.

Sinocalanus doerrii, an introduced freshwater calanoid copepod, was first recorded in spring 1979 and was most abundant in summer and fall during the early 1980s (Figure 4). Spring 2004 abundance increased from 2003; but there is no visible long-term trend. Long-term declines occurred in summer and fall, culminating in the lowest abundance in the mid 1990s. Abundance then increased with the greatest increases occurring in summer. However, in 2004, both summer and fall indices were among the lowest recorded.

The genus *Acanthocyclops* includes several native freshwater cyclopoid copepod species. They have experienced consistent downtrends in all seasons since the 1970s (Figure 5), with summer abundance well below the long-term trend line from 1989-1995. In 2004, abundance decreased slightly in spring but increased in summer and fall relative to 2003.

The most abundant cladoceran in the upper estuary are the native *Bosmina*, *Daphnia*, and *Diaphanosoma*, all freshwater genera. Combined they have shown an overall downward trend in all seasons since the early 1970s, especially in fall (Figure 6), although spring and summer abundance have been relatively stable since the late

1980s. In 2004 abundance increased in spring and summer, but decreased slightly in fall compared to 2003. There appears to be a trend of increased abundance for the last 3 or 4 years in all seasons.

The native brackish water rotifer, *Synchaeta bicornis*, is most abundant in summer and fall (Figure 7). Its spring abundance pattern has been erratic, but declining; none have been collected in spring since 2001. Its summer and fall abundance have shown long-term declines since the late 1970s. In 2004, summer abundance increased from 2003 while fall abundance decreased. Abundance of all rotifers except *S. bicornis* declined from the early 1970s through the 1980, but stabilized since the early 1990s (Figure 7). The few periodic declines since 1990 were greatest in summer. In 2004, spring abundance was down from 2003, while summer abundance was essentially the same and fall abundance increased slightly.

The introduced mysid, *Acanthomysis bowmani*, has been the most abundant mysid in the estuary since fall 1993, when it was first recorded (Figure 8). However, it has not been as abundant as *Neomysis mercedis* was in the 1970s through mid 1980s. Its abundance increased in spring and fall 2004, but declined somewhat for summer.

Spring and summer indices have been relatively stable since 1998 but fall abundance has been declining.

Neomysis kadiakensis began appearing regularly in the macro-zooplankton catches in 1996. It is now the second most abundant mysid in the upper estuary. Recent evidence suggests that the species we've identified as *N. kadiakensis* may be an introduced species known as *N. japonica* (John Chapman, personal communication). Its abundance has slowly increased in all seasons (Figure 8). It is most abundant in spring, when its range extends almost into fresh water as far upstream as the confluence of the Sacramento and San Joaquin Rivers. Spring and summer 2004 abundance indices were lower than for 2003 while the fall index was higher.

Neomysis mercedis was once the only common mysid in the upper estuary, were it served as an important food source for fish. It suffered a population collapse in all seasons in 1989 (Figure 8). This was probably caused by predation and competition from *Corbula amurensis*. Spring and fall 2004 *N. mercedis* abundance increased from 2003, but the summer index decreased. The indices for all 3 seasons remained very low.
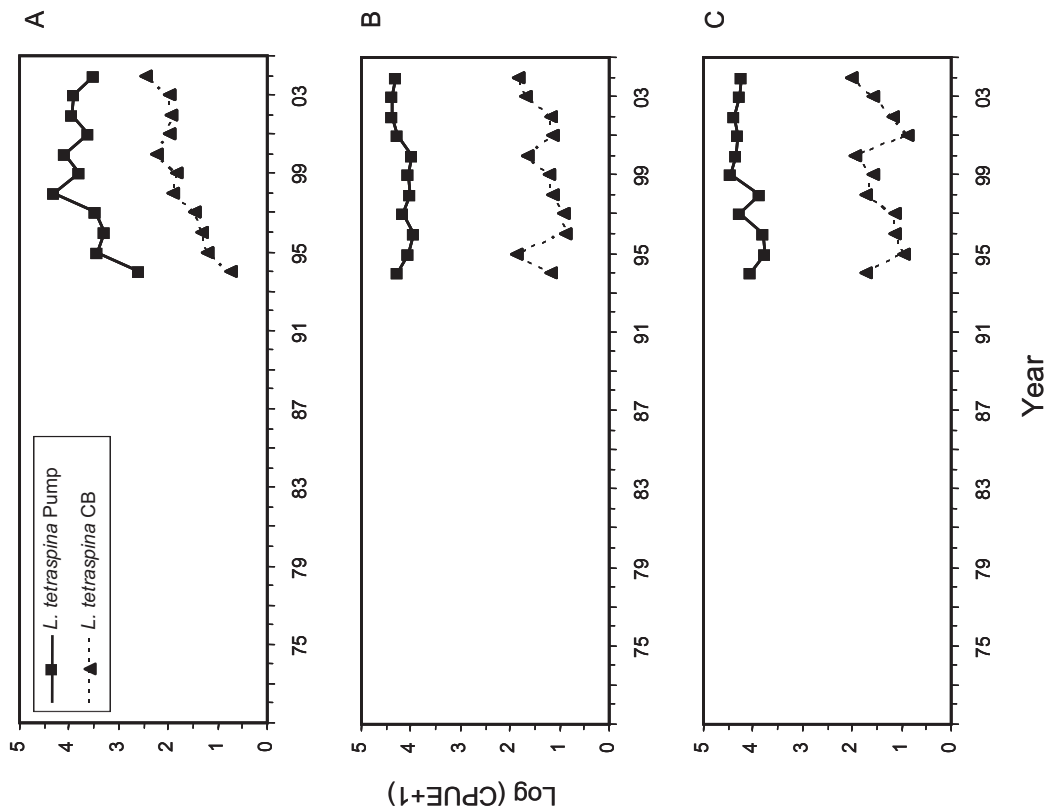

Figure 1. Log of mean abundance of *Limnoithona tetraspina* from the pump and CB net in Spring (A), Summer (B), and Fall (C), 1993 – 2004.

Figure 3. Log of mean abundance of *Acartia* spp. and *Acartiella sinensis* from the CB net in Spring (A), Summer (B), and Fall (C), 1972 – 2004.
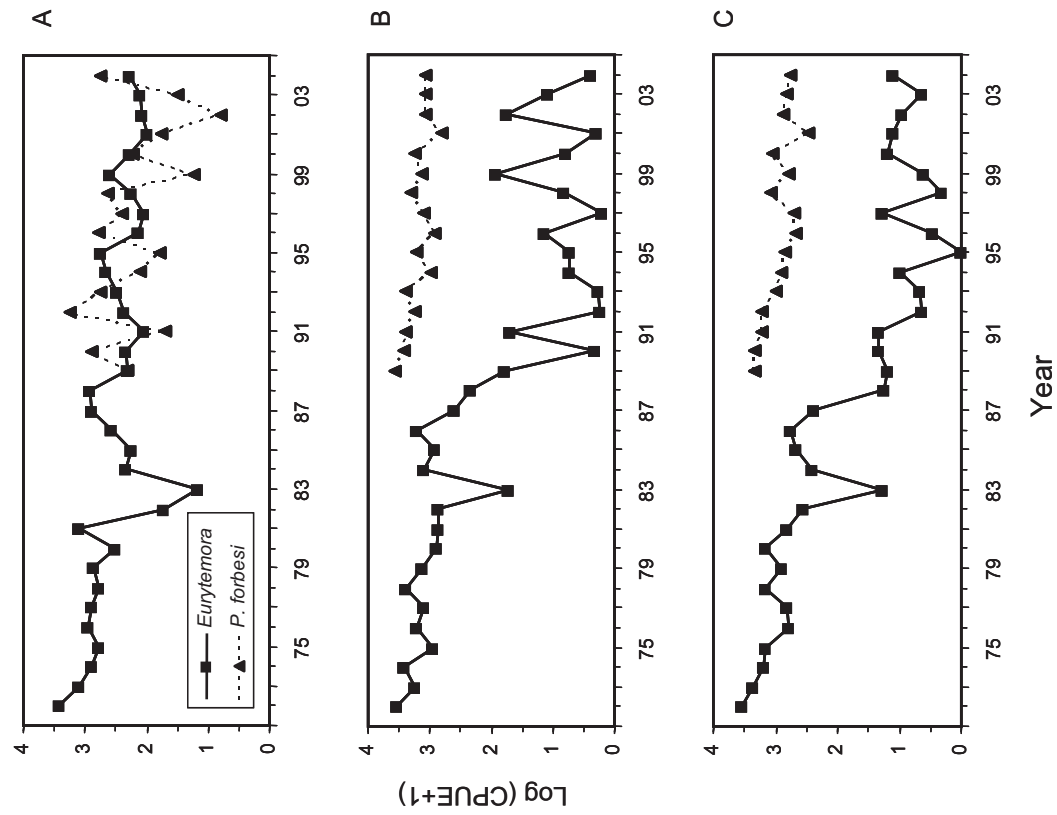
Figure 2. Log of mean abundance of *Eurytemora affinis* and *Pseudodiaptomus forbesi* from the CB net in Spring (A), Summer (B), and Fall (C), 1972 – 2004..
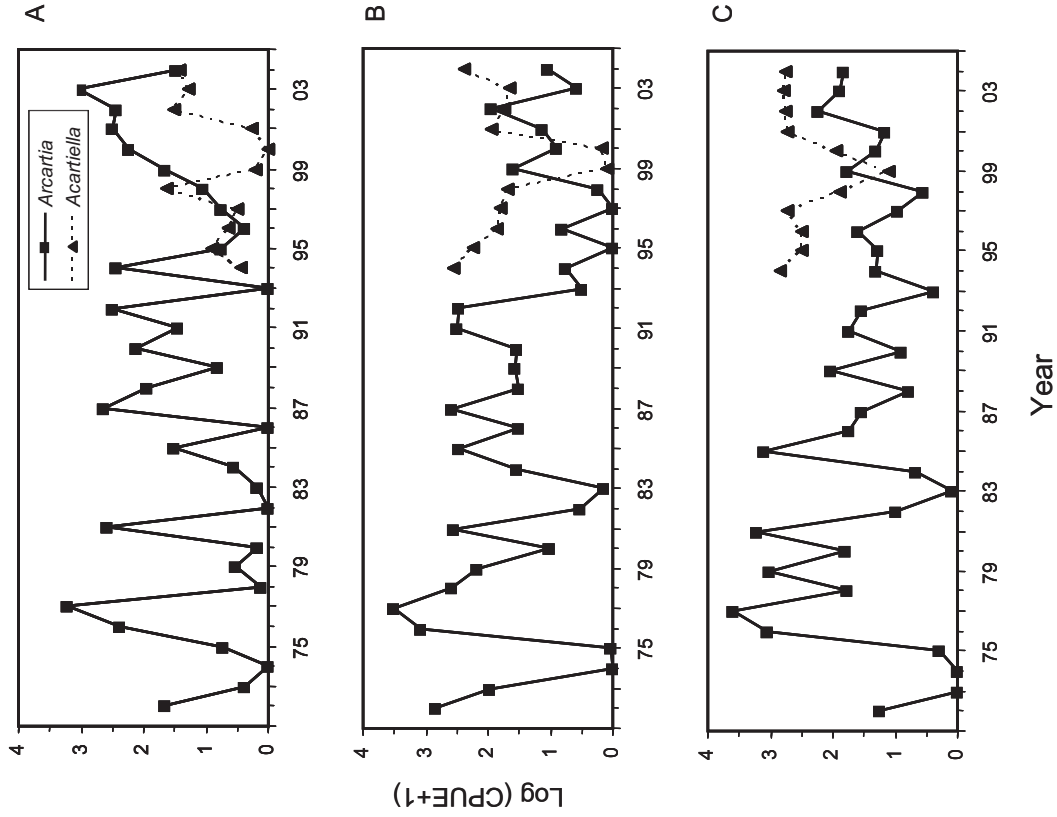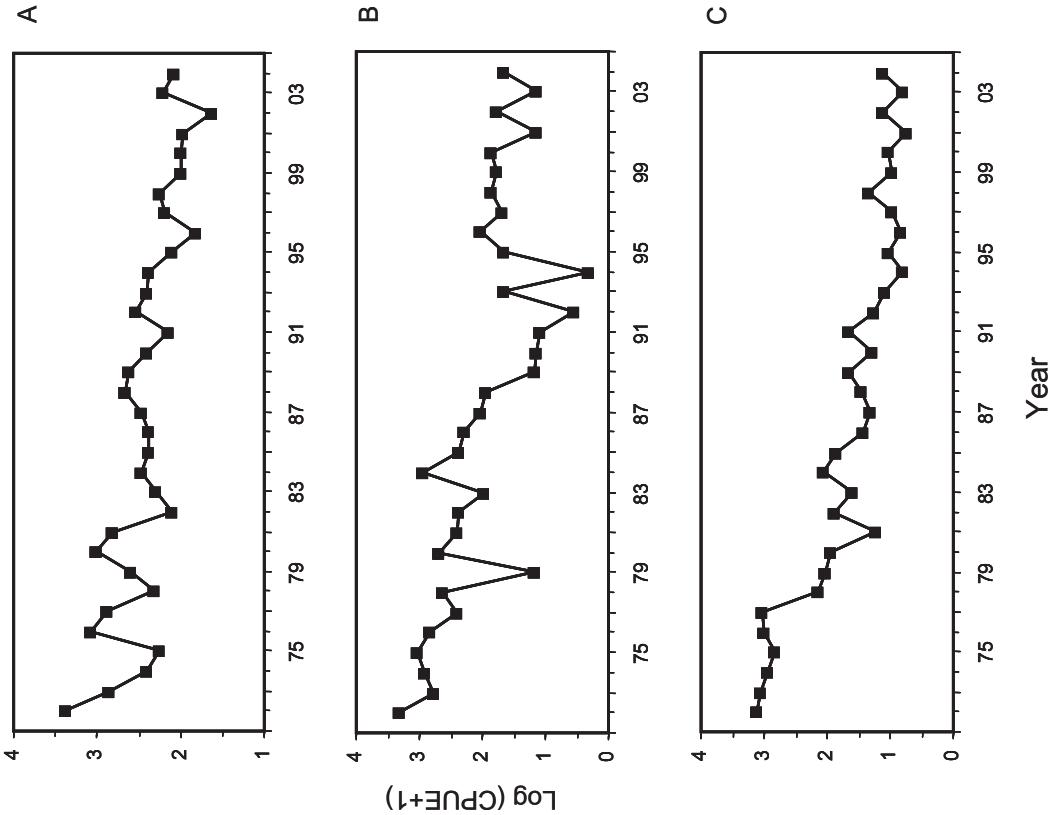
Figure 5. Log of mean abundance of *Acanthocyclops* spp from the CB net in Spring (A), Summer (B), and Fall (C), 1972 – 2004.

Figure 4. Log of mean abundance of Diaptomidae and *Sinocalanus doerrii* from the CB net in Spring (A), Summer (B), and Fall (C), 1972 – 2004.

Figure 7. Log of mean abundance of *Synchaeta bicornis* and rotifers excluding Synchaeta bicornis from the pump in Spring (A), Summer (B), and Fall (C), 1972 – 2004.

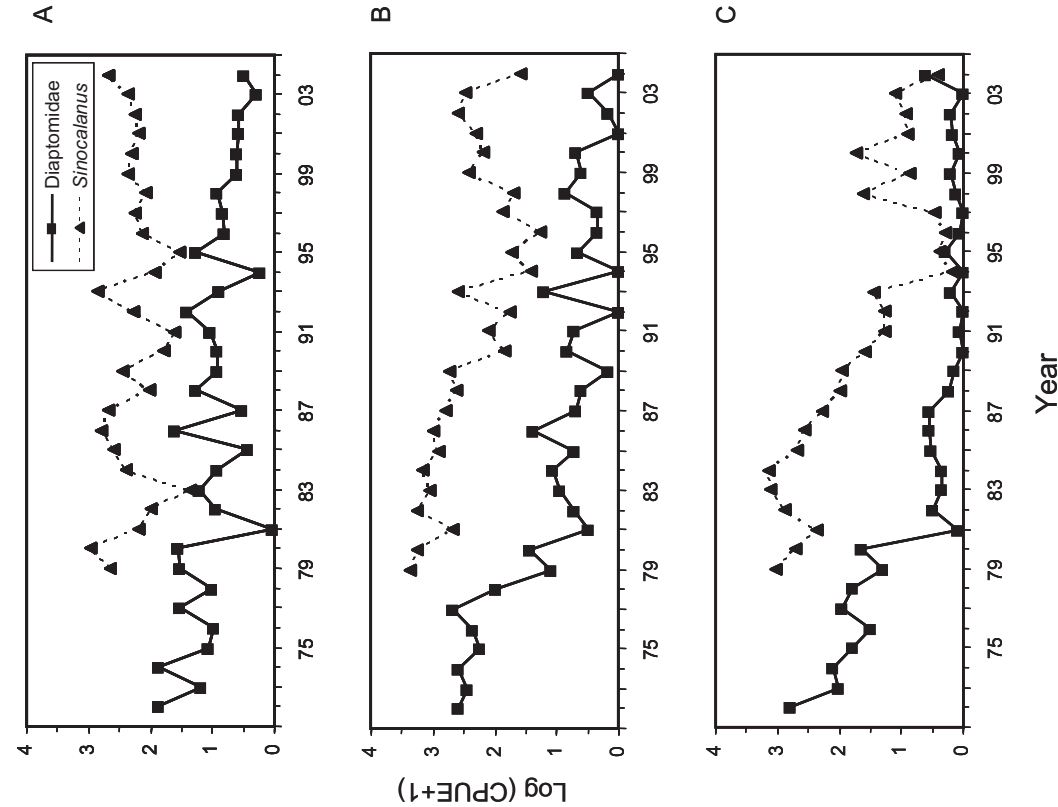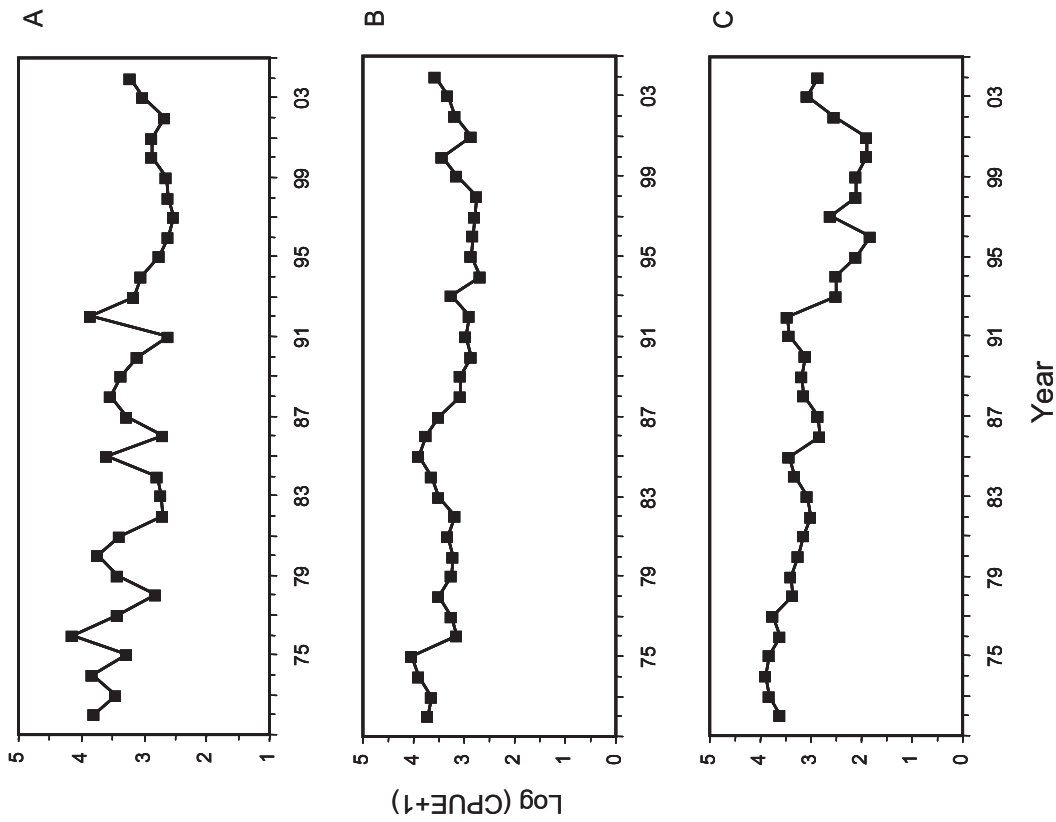Figure 6. Log of mean abundance of Cladocera from the CB net in Spring (A), Summer (B), and Fall (C), 1972 – 2004.
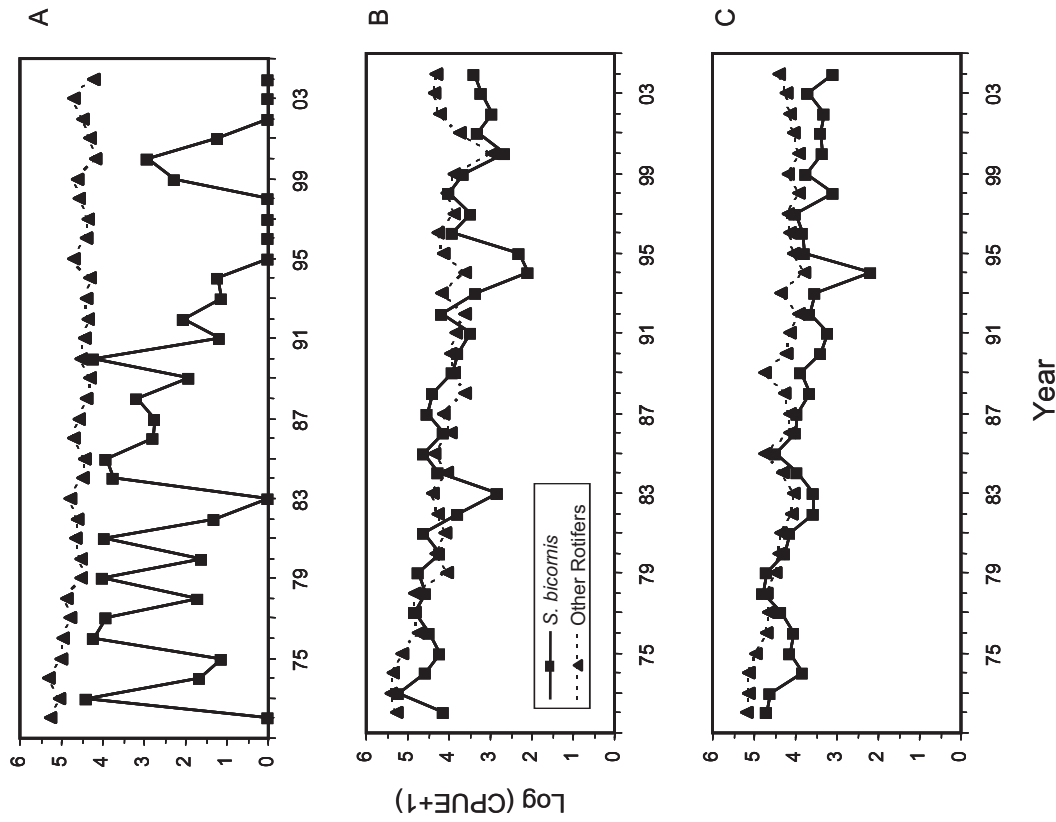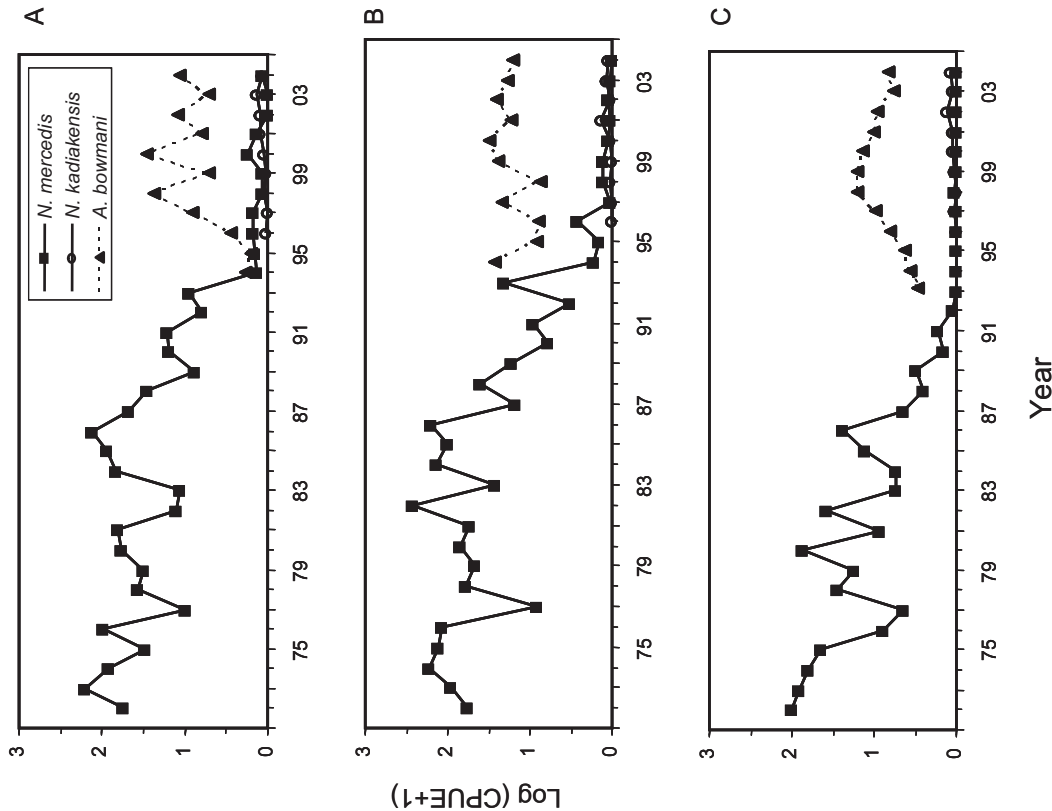
Figure 8. Log of mean abundance of *Neomysis mercedis*, *Acanthomysis bowmani*, and *Neomysis kadiakensis* in Spring (A), Summer (B), and Fall (C), 1972 – 2004.

Common Crabs of the San Francisco Estuary

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This report summarizes the abundance trends and distributional patterns of the most common *Cancer* crabs and the Chinese mitten crab in the San Francisco Estuary. Most of the data is from the San Francisco Bay Study (Bay Study) otter trawl, with additional mitten crab data from Suisun Marsh trawls and CVP and SWP salvage.

Cancer crabs

Cancer magister, the Dungeness crab, is a valuable sport and commercial species that reproduces in the ocean in winter and rears in nearshore coastal areas and estuaries. Small juvenile *C. magister*, 5-10 mm carapace width (CW), immigrate to San Francisco Estuary in spring, rear for 8-10 months, and emigrate to the ocean in fall and win-

ter at a size of approximately 100 mm CW. Estuary-reared crabs reach legal size at the end of their third year, one to two years before ocean-reared crabs.
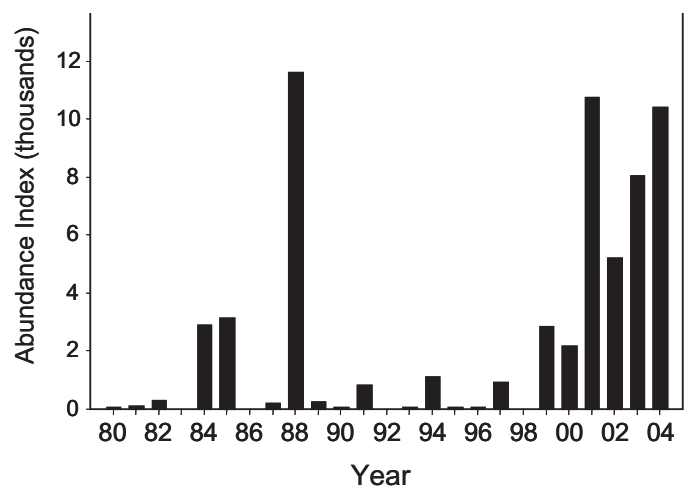

Figure 1 Annual abundance of age-0 Cancer magister, Bay Study otter trawl, May-July. 1980-2004.

The abundance index of age-0 *Cancer magister* increased slightly in 2004 from 2003 and was the third highest index for the 1980-2004 study period (Figure 1). With the strong 2004 year class, four of the five highest age-0 *C. magister* indices were in the past four years. We believe that these high indices were a result of cooler ocean temperatures, which result in increased embryo and larval survival, and a moderate or weak northward Davidson Current, which results in retention of larvae near shore and the Golden Gate. Although SSTs were slightly above average in winter 2003-04, nearshore temperatures were cool compared to the strong El Niño years of the 1980s and 1990s. Also, the Davidson Current was moderate when compared to years with very frequent winter storms, such as 1982, 1983, 1995, and 1998.

These recent strong year classes have been reflected in the Central California *Cancer magister* crab landings. Landings have surpassed 4 million pounds the past three fishing seasons, with 5.2 million pounds landed in the 2004-05 season through early February 2005; Central California landings last exceed 4 million pounds in the late 1950s. The 2001 year class of estuary-reared crabs was legal size and available to the fishery in the 2003-04 season, the 2002 year class in the 2004-05 season, and the 2003 year class will be available in the 2005-06 fishing season.

In 2004, the first age-0 *Cancer magister* were collected in April, with the highest abundance in May. Catches were initially highest in the Central Bay channel, then in San Pablo Bay and Carquinez Strait as the crabs moved upstream. Similar to other recent years with high abundance, we continued to collect smaller age-0 *C. magister* at our Alcatraz Island station throughout summer and fall. Although these smaller crabs are not rearing in the estuary for their entire first year, they do contribute to the annual abundance index.

Cancer antennarius, the brown rock crab, is common to rocky areas and other areas with structure. It and *C. productus*, the red rock crab, are targeted by sport anglers fishing from piers and jetties in the higher salinity areas of the estuary. The abundance of age-0 *C. antennarius* increased dramatically in 2004, and was the highest for the study period (Table 1). This continued the trend of above average indices since the mid-1990s. This large year class probably settled in May and June, as 79% (n=142) of crabs 5-9 mm CW was collected in June and July. In 2004, age-0 *C. antennarius* were collected from

South Bay through San Pablo Bay, but were most common at shoal stations in South Bay near Alameda, in Central Bay near the Berkeley Pier, and in San Pablo Bay near Point Pinole. There was little indication of movement or migration over the year. From the distribution of age-0 *C. antennarius,* we concluded that most hatched in the Bay rather than the ocean.

Table 1 Annual abundance indices of age-0 *Cancer* crabs from the Bay Study otter trawl, 1980-2004. The index period is May-October for all species.

Year	C. gracilis age-0	C. antennarius age-0	C. productus age-0
1980	17	102	0
1981	152	76	6
1982	87	0	4
1983	151	28	4
1984	154	50	41
1985	216	20	38
1986	59	0	89
1987	93	71	79
1988	223	21	138
1989	203	29	30
1990	159	113	160
1991	656	171	128
1992	371	60	62
1993	616	398	71
1994	1017	603	166
1995	227	367	40
1996	411	1126	198
1997	1131	351	86
1998	1621	718	149
1999	222	90	249
2000	251	849	93
2001	1921	276	142
2002	796	119	238
2003	522	424	140
2004	112	1765	139

Cancer gracilis, the slender crab, is smaller than the other 3 species of *Cancer* crabs, rarely exceeding 85 mm CW. It is common in open sandy or sand-mud habitats rather than rocky areas; researchers have hypothesized that because of its size it cannot compete with the rock crabs for the more "preferred" protected habitats. In contrast to *C. magister* and *C. antennarius*, the abundance of

age-0 *C. gracilis* decreased in 2004 (Table 1). This followed a decade plus of relatively high indices. Age-0 *C. gracilis* were collected from South Bay through San Pablo Bay, although the San Pablo Bay collections were spo-radic. The highest catches were at channel stations in Central Bay near Yerba Buena Island and Angel Island.

Table 2 Annual adult *Eriocheir sinensis* CPUE and estimated total salvage, 1996-2004. Bay Study CPUE is October$_{year}$-March$_{year +1}$, Suisun Marsh is July-December, and CVP and SWP salvage is September-November.

Year	Bay Study CPUE (#/tow)	Suisun Marsh CPUE (#/tow)	CVP salvage est. total	SWP salvage est. total
1996	0.02	0.00	50	
1997	0.34	0.07	20000	
1998	2.51	0.89	750000	
1999	0.96	1.08	90000	34000
2000	0.93	0.02	2500	4700
2001	3.25	0.17	27500	7300
2002	1.07	0.04	2400	1200
2003	0.15	0.00	650	90
2004	0.12	0.00	750	370

Cancer productus, the red rock crab, is the least common of the 4 *Cancer* crabs we usually collect in the estuary, reflecting a strong preference for rocky intertidal and subtidal marine habitats rather than its actual abundance. The 2004 abundance index of age-0 *C. productus* was almost identical to the 2003 index (Table 1). In 2004, 63% (n=32) of the age-0 *C. productus* were collected at our Alcatraz Island station, which has a substrate of cobble and small rocks.

Chinese mitten crab

Eriocheir sinensis, the Chinese mitten crab, was first collected in the estuary in the early 1990s, but likely introduced to South Bay in the late 1980s. After several years of rapid population growth and expanding distribution, the population of *E. sinensis* peaked in 1998-99 (Table 2). All data sources indicate that the population has been low the past three years. In 2004, the Bay Study adult *E. sinensis* mean CPUE was the lowest since 1996, the first year we collected it north of the Bay Bridge. Suisun Marsh adult CPUE was again zero in 2004, although several juvenile crabs were collected early in the year. The 2004 CVP and SWP estimated total salvage was approximately 1,100 crabs, slightly higher than in 2003 but still very low relative to 1998-99

Also, public reports of *E. sinensis* sightings in 2004 to the toll-free reporting line (1-888-321-8913) dropped to the lowest level (4 reports) since the line was established in 2001. Passive habitat trapping by USFWS at 13 sites distributed throughout the Sacramento-San Joaquin Delta detected zero *E. sinensis* between July and October 2004 (Bergendorf 2005, personal communication, see notes).

The downstream migration of adult *E. sinensis* was typical of a low abundance year, with the first adult collected at the CVP in early September and at the SWP the third week of September. Peak migration was early October at the CVP and late October at the SWP. The Bay Study first collected adult crabs in the lower Sacramento and San Joaquin rivers in October. Migration further downstream was very slow, with most crabs collected in Suisun and Honker bays through December. The first crab was collected in San Pablo Bay in January 2005; increased outflow resulted in a movement of almost all adult *E. sinensis* to San Pablo Bay by March 2005.

Acknowledgements

Notes
David Bergendorf, USFWS, e-mail, May 5, 2005.

Fishes in the San Francisco Estuary, 2004 Status and Trends

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Introduction

The 2004 Status and Trends fishes report includes data from 4 of IEP's long-term monitoring surveys in the San Francisco Estuary: 1) the Summer Townet Survey (TNS), 2) the Fall Midwater Trawl Survey (FMWT), 3) the San Francisco Bay Study (Bay Study), and 4) the Delta Smelt 20-mm Survey (20-mm Survey). The most recent abundance indices, long-term abundance trends, and distributional information are presented for the most common species in the estuary and some less-common species of interest, such as splittail and some of the surfperches. Several pelagic species that spawn and rear in the upper estuary have undergone severe declines in recent years and are presented first. This group is followed by the upper estuary demersal fishes, the marine pelagic fishes, surfperches, and marine demersal fishes. Within each section, the species are presented phylogenetically.

Methods

The TNS has been conducted annually since 1959, except for 1966, 1983, and 2002. It produces annual abundance indices for age-0 striped bass (= the 38-mm index) and age-0 delta smelt. The TNS begins in June and samples 32 sites from eastern San Pablo Bay to Rio Vista on the Sacramento River and Stockton on the San Joaquin River. Historically the number of surveys ranged from 2 to 5 each year; as of 2003, it was standardized to 6 surveys per year. The striped bass index is interpolated between the 2 surveys that bracket the mean size of 38.1-mm fork length (FL) (Chadwick 1964, Turner and Chadwick 1972). The delta smelt index is the average of the first 2 survey indices. The 2004 TNS completed 6 surveys at 2 week intervals from June 14 to August 27, 2004.

The FMWT has sampled annually since 1967, with the exception of 1974 and 1979. It was designed to determine the relative abundance and distribution of age-0 striped bass in the estuary, but data is also used for other upper estuary pelagic species, including American shad, delta smelt, and longfin smelt. The FMWT survey samples 116 stations monthly from September to December in an area ranging from San Pablo Bay to Stockton on the San Joaquin River and Hood on the Sacramento River. The index calculation (Stevens 1977) uses catch data from 100 of the 116 stations; the remaining 16 stations increase spatial coverage for delta smelt.

The Bay Study has sampled from South San Francisco Bay to the western delta monthly with an otter trawl and midwater trawl since 1980. There are a few data gaps, most significantly limited sampling with the midwater trawl in 1994 and no winter sampling from 1989 to 1997. Abundance indices are routinely calculated for 35+ pelagic and demersal fishes and several species of crabs and caridean shrimp; only the most common species are included in this report. The Bay Study samples 52 stations, of which 35 have been consistently sampled since 1980 and are used for the abundance indices. Additional information about the study, including index calculation, can be found in IEP Technical Report 63 (Baxter et al. 1999).

The 20-mm Survey monitors larval and juvenile delta smelt distribution and relative abundance throughout their historical spring range, which includes the entire delta downstream to San Pablo Bay and the Napa River. Surveys have been conducted every other week from early March through July since 1995. Three tows are completed at each of the 48 stations with a 1,600 μm mesh net (Dege and Brown 2004). This survey gets its name from the size (20 mm) at which delta smelt are retained and readily identifiable at the CVP and SWP fish facilities.

Data from all 4 surveys was used to describe trends and distribution of upper estuary pelagic fishes when available, but only Bay Study midwater trawl data was used for the marine pelagic fishes and Bay Study otter trawl data for demersal fishes.

Physical Setting

The 2004 winter-spring delta outflow was slightly higher than 2003 and approximately double that of either 2001 or 2002. Although the mean daily January-May outflow of 1072 cms was the highest since 2000, water year 2004 was classified as "Below Normal" based on the Sacramento Valley Index and "Dry" based on the San Joaquin Valley index. There were 2 peak outflow events in 2004,

one of approximately 2,000 cms in early January and another of approximately 5,000 cms in early March (see Kate Le's article in this issue).

The San Francisco Estuary is situated between 2 major faunal regions, the cold-temperature fauna of the Pacific Northwest and the subtropical fauna of southern and Baja California, and as such is a transitional area with elements of both faunas (Parrish et al. 1981). The northern Pacific Ocean has been in a cold-water regime since 1999 (Peterson and Schwing 2003), which is hypothesized to be beneficial to many cold-temperate species, including Dungeness crab, English sole, and many of the rockfishes.. However, a weak El Niño event resulted winter 2003-2004 sea surface temperatures (SSTs) that were slightly warmer (<0.5 °C) and summer SSTs that were 1-2 °C warmer than the historic means in the Gulf of the Farallones. Concurrent with these warmer summer SSTs, coastal upwelling near San Francisco Estuary was relatively strong in May and June but weakened in July and August; overall, summer upwelling was weaker in 2004 than in 2003.

Upper Estuary Pelagic Fishes

American shad

The American shad (Alosa sapidissima) is an introduced anadromous species that spawns in the rivers in late spring, rears in fresh water through summer, and migrates to the ocean in late summer and fall. It rears for 2-5 years in the ocean before returning to fresh water to spawn. Most males mature at age 3 or 4 and most females at age 4 or 5; many fish only spawn once, but some fish spawn annually, reaching a maximum age of 7 years. All life stages of American shad are planktivores.

The FMWT 2004 American shad index was 10% of the 2003 index, which happened to be the highest index for the study period (Figure 1A). With the exception of 2003, American shad abundance has been relatively low since 1999, reversing the trend of increasing abundance through 1998. American shad were also not nearly as widespread in 2004 as in 2003.
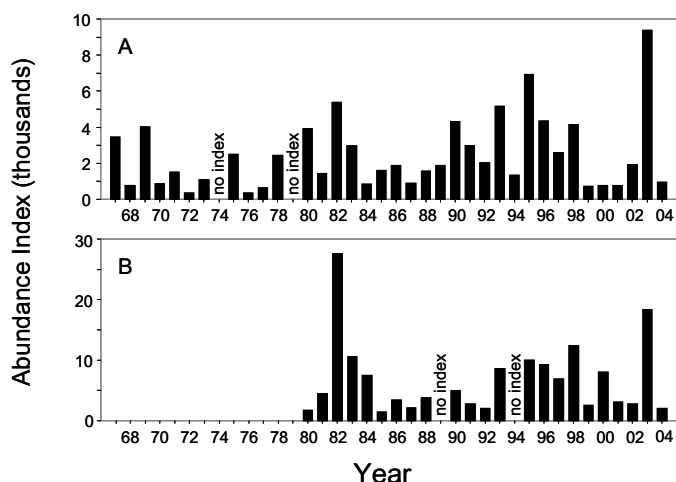

Figure 1 Annual abundance of American shad: A) FMWT, all sizes, September-December. B) Bay Study midwater trawl age-0, July-October

The Bay Study 2004 age-0 American shad index was only 11% of the previous year's index (Figure 1B); however, the 2003 index was the second highest for the study period. The 2004 index was also the fourth year since 1998 of below average indices. American shad were initially collected only at river stations in July, but were collected throughout Suisun and San Pablo bays in fall as they migrated to sea.

Threadfin shad

The threadfin shad (*Dorosoma petenense*) is a small, short-lived introduced species that is also a planktivore at all life stages. It reproduces in freshwater but can be found throughout the estuary. In river systems, it is most common in slower moving waters, such as dead-end sloughs. The 2004 FMWT threadfin shad index was very low for the third consecutive year (Figure 2). This is a drastic change from recent years, as 3 of the 5 years prior to 2002 had record high indices. In all months, the majority of threadfin shad was collected in the lower San Joaquin River and the eastern Delta, especially in the Stockton Deep Water Channel between the Calaveras River and Fourteenmile Slough. However, distribution expanded westward in December, when 45% of the index was from stations in San Pablo Bay, Carquinez Strait, and Suisun Bay.
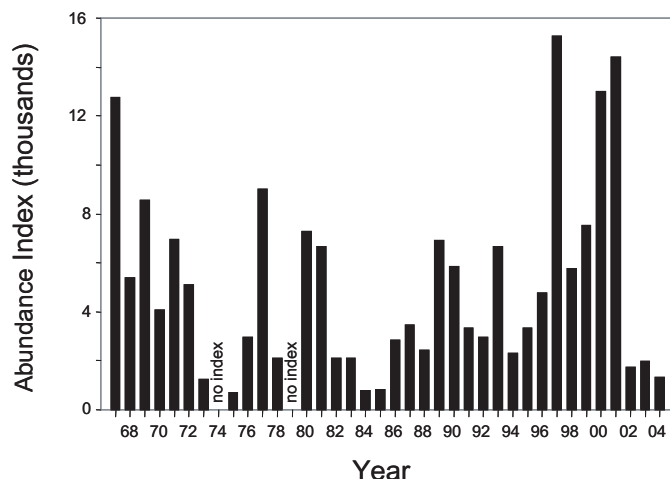
Figure 2. Annual abundance of threadfin shad, FMWT, September-December.

Delta smelt

The 2004 TNS age-0 delta smelt index was 2.9, a slight increase from the 2003 index of 1.6 (Figure 3A). The 2004 index was driven by the catch of 65 fish from the station near Sherman Island on the Sacramento River in Survey 1, representing 87% of the total delta smelt collected in surveys 1 and 2, which comprise the index. The majority of delta smelt was collected in the Sacramento River and Suisun Bay (Table 1), with a few collected in the San Joaquin River in surveys 2, 3, and 4.

The 2004 FMWT delta smelt abundance index was 35% of the 2003 index and the lowest index on record (Figure 3B), but only slightly lower than the 1985 and 1994 indices. Distribution did not change over the sampling period, as delta smelt were collected only in Suisun Bay and the lower Sacramento River during 2004. The percentage of the FMWT delta smelt index from the lower Sacramento River never fell below 62%, which is a common distribution pattern for this species in the FMWT.

Contrary to TNS and FMWT indices, the 2004 Bay Study midwater trawl age-0 delta smelt index was the 6th highest for the study period (Figure 3C). This index was driven by 2 large catches at the Sacramento River station near Sherman Island, where 29 fish were collected in September and 16 fish in October. The 45 fish collected in these 2 tows accounted for 88% of all delta smelt collected by the Bay Study midwater trawl in 2004. Omitting 2004, the overall trend in recent years is one of low indices, with

the 2nd and 3rd lowest indices for the study period in 2001 and 2002.

The 2004 20-mm Survey delta smelt index was 10.4 (n=649 fish), the 2nd lowest for the study period (Figure 3D). This was the 4th consecutive year of relatively low delta smelt abundance in this survey. Larval and juvenile delta smelt (combined mean size 9.5 mm TL) were first observed during survey 1 downstream of the confluence and in Montezuma Slough and the south Delta. By mid May, larger larvae and juveniles (combined mean size 19.4 mm FL) were concentrated from the lower San Joaquin and Sacramento Rivers to Montezuma Slough. During the last 2 surveys, most fish were juveniles (mean size 34.5 mm FL) and were concentrated in the lower Sacramento River with a limited distribution in Suisun Bay and Cache Slough. This delta smelt distribution is typical of a below-normal outflow year, when river outflow and tidal movement concentrate fish in the lower Sacramento River and the confluence.

Table 1 Percentages of Townet Survey index by area for striped bass and delta smelt for surveys 1-6, 2004.

| Species and Area	Survey 1	Survey 2	Survey 3	Survey 4	Survey 5	Survey 6
Striped Bass						
Montezuma Slough	29.0	11.0	3.0	100.0	100.0	0.0
Suisun Bay	16.0	24.0	35.0	0.0	0.0	0.0
Sacramento River	39.0	23.0	53.0	0.0	0.0	0.0
San Joaquin River	8.0	40.0	9.0	0.0	0.0	0.0
East Delta	5.0	2.0	0.0	0.0	0.0	0.0
South Delta	3.0	0.0	0.0	0.0	0.0	0.0
Delta Smelt						
Montezuma Slough	0.6	0.0	0.0	0.0	0.0	0.0
Suisun Bay	0.2	27.9	27.5	82.1	62.3	100.0
Sacramento River	99.1	66.0	59.9	15.4	37.7	0.0
San Joaquin River	0.0	6.1	12.6	2.5	0.0	0.0
East Delta	0.0	0.0	0.0	0.0	0.0	0.0
South Delta	0.0	0.0	0.0	0.0	0.0	0.0
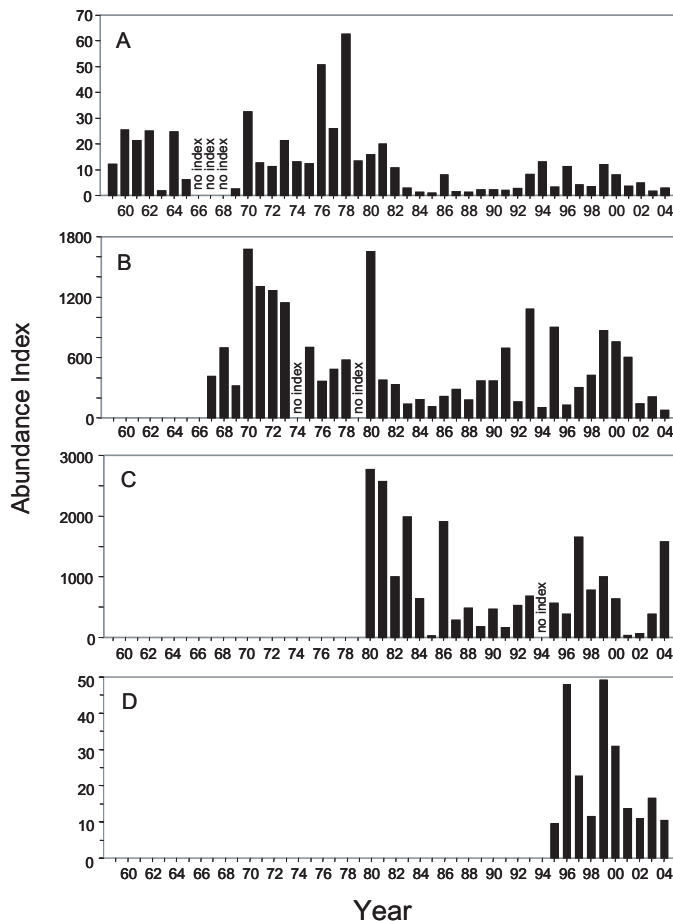
Figure 3 Annual abundance of delta smelt: A) TNS age-0; B) FMWT, all sizes, September-December; C) Bay Study midwater trawl age-0, June-October; D) 20-mm Survey larvae and juveniles

Water temperatures were above average in late winter and early spring 2004; mean delta water temperatures were approximately 17.4°C by the 3rd week of March, nearing the upper limit (18-20°C) for delta smelt spawning (Figure 4). The spawning season was largely constrained between March and April and resulted in 2 cohorts, a smaller one in March, and a larger one in April. Most of the 2004 year-class production occurred over a short time period, which raised concern among IEP biologists that the species was more susceptible to a catastrophic event. The low abundance measured by the 20-mm Survey and later by the TNS continued to the fall months, as the FMWT recorded their lowest age-0 delta smelt index in 38 years.
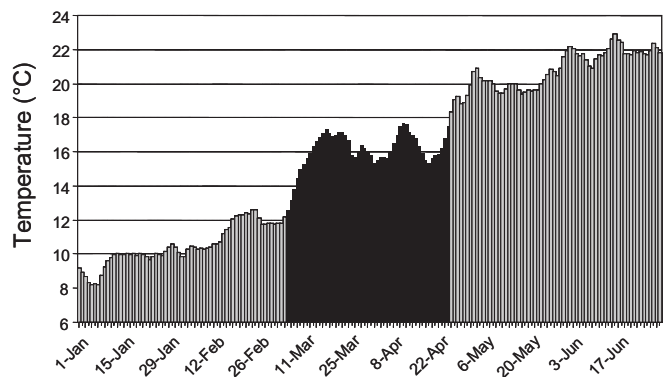

Figure 4. Daily average delta water temperatures measured from Antioch, Mossdale, and Rio Vista (January 1 - June 30, 2004). The 2004 delta smelt spawning window indicated by solid black bars.

Although CVP and SWP delta smelt salvage never triggered "yellow light" or "red light" conditions under the 1995 USFWS Biological Opinion in 2004, concern regarding low abundance resulted in a Delta Smelt Working Group recommendation that Environmental Water be used to reduce exports for 2 weeks post-"VAMP".

Longfin smelt

The longfin smelt (*Spirinchus thaleichthys*) is a short-lived anadromous species that spawns in freshwater in late winter and spring and rears in fresh to brackish water. Some age-0 and age-1 fish apparently emigrate to the ocean in late-summer and fall for a short period, often returning to the estuary in late fall of the same year. A few longfin smelt mature at the end of their first year and the remainder at the end of the 2nd year, with a few living to spawn again at age-3. A strong positive correlation between longfin smelt abundance and outflow has been reported (Stevens and Miller 1983). However, this relationship changed in the late 1980s - there is still a correlation between abundance and outflow, but abundance is now at a lower level relative to outflow (Kimmerer 2002). Possible reasons for this change include a decline in phytoplankton and zooplankton abundance due to grazing by the introduced clam, *Potamocorbula amurensis* (Kimmerer 2002), and dominance of the introduced copepod, *Limnoithona tetraspina*, which is very small and may not be consumed by larval and juvenile fishes.

The 2004 FMWT longfin smelt abundance index was almost identical to the 2003 index, and the 3rd lowest

index on record (Figure 5A). The only other years with lower indices were 1991 and 1992, coincident with the end of a protracted drought. Consistent with recent years, the FMWT collected longfin smelt from San Pablo Bay to the lower Sacramento River, with the center of distribution in Suisun Bay.

The 2004 Bay Study age-0 longfin smelt abundance index increased from 2003 for the midwater trawl (Figure 5B) and decreased slightly for the otter trawl (Figure 5C). Although the 2004 midwater trawl index increased, 2004 was the 5th consecutive year of low indices and the 8th lowest index for the study period. The 2004 otter trawl longfin smelt index also marked the 5th consecutive year of relatively low indices. Although longfin smelt had a modest recovery from 1995 to 1999, the mean 2000-2004 midwater trawl index was only 3% of the mean 1995-1999 index. The decline in the otter trawl was not as precipitous, as the mean 2000-2004 index was 17% of the 1995-1999 mean.

The Bay Study first collected a few age-0 longfin smelt in April in San Pablo Bay. Catches increased from June through August and fish were distributed from Central Bay to the lower Sacramento River at shoals and channels. In September and October catches decreased and distribution contracted, with most fish collected in Central Bay and the channels of San Pablo and Suisun bays. In November and December, catches increased and distribution expanded, with fish collected from South Bay to the lower Sacramento River. Trace element analysis of longfin smelt otoliths found fish initially rearing in brackish waters and moving either into low salinity or marine waters early in their first year of life, indicative of alternate life history strategies (Hobbs et al. 2005).
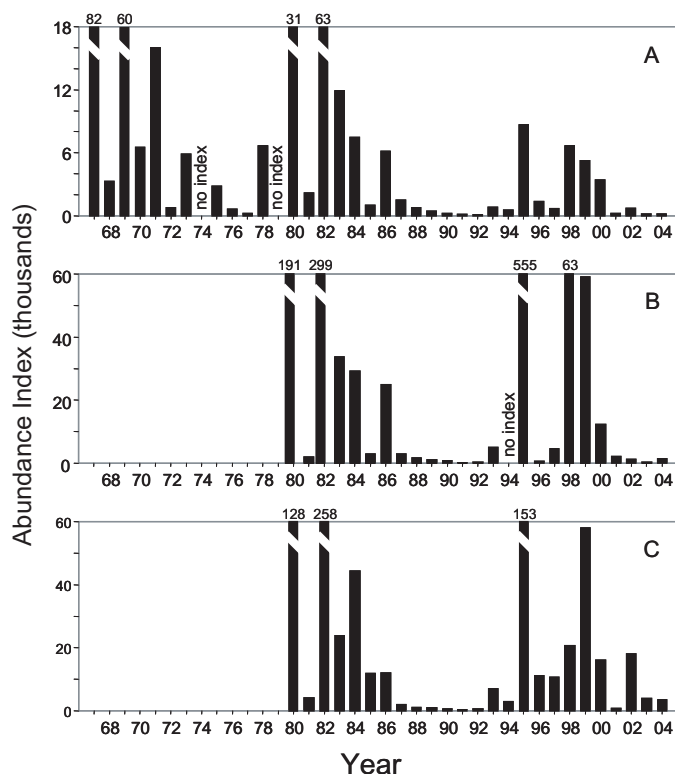

Figure 5 Annual abundance of longfin smelt: A) FMWT, all sizes, September-December; B) Bay Study midwater trawl age-0, May-October; C) Bay Study otter trawl age-0, May-October

Splittail

The splittail (*Pogonichthys macrolepidotus*) is endemic to the San Francisco Estuary and its watershed. It spawns in late winter and spring in freshwater areas, most notably floodplains, and rears in fresh and brackish water from near Colusa on the Sacramento River to San Pablo Bay. Splittail also inhabit the Napa and Petaluma rivers and associated marshes. Males mature the end of their 2nd year, females at the end of their 3rd year, and fish live to at least 8 years. In late February and early March 2004, river flows increased sufficiently to inundate river terraces and floodplain, but inundation did not last the 21-30 days necessary for spawning, egg incubation, and larval development to swimming competence that result in strong year-classes. River levels remained below floodplain level for most of the March-May spawning period, resulting in suboptimal spawning and rearing conditions. Although a 2004 year class was detected by all surveys, indices were low.

The 2004 FMWT splittail index (all ages combined) was 4, continuing the trend of low indices since 1998

(Figure 6A). The age-0 index was 3 and the age-1+ index was 1. The index was comprised of 4 fish that were collected at 3 stations in Suisun Bay, Grizzly Bay, and Montezuma Slough. The 2004 Bay Study age-0 splittail midwater trawl index increased slightly from 2003 (Figure 6B). As for the FMWT, the last strong year class was in 1998. This large year class is evident in the Bay Study's age-1 and age-2+ indices, with the largest age-1 index in 1999 (Figure 6C) and the largest age-2+ index in 2000 (Figure 6D). We collected 12 age-0 splittail in 2004, but only 6 were from index stations. Eight were collected in Suisun and Honker bays and 4 in the Sacramento River near Rio Vista.
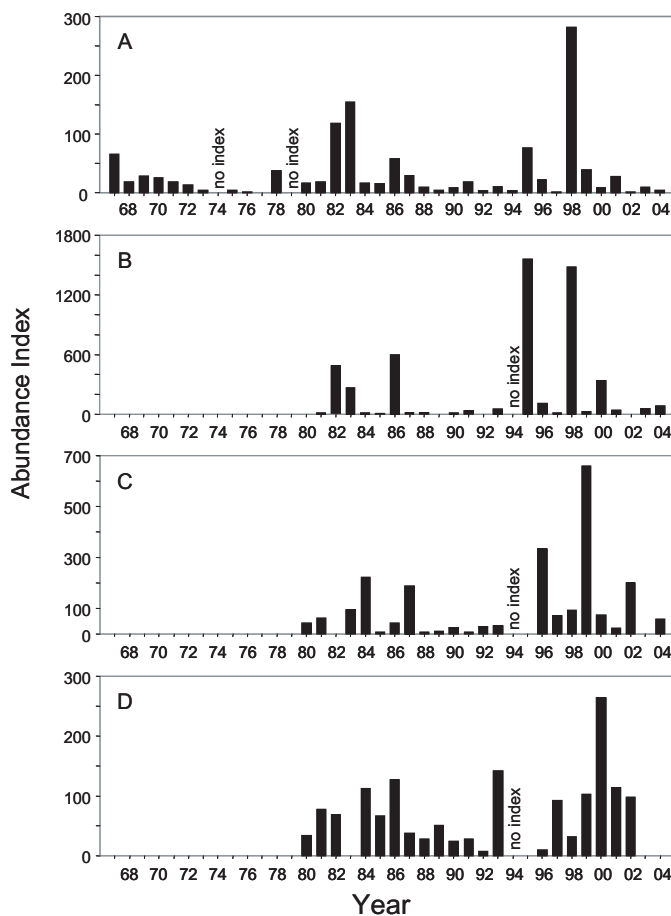

Figure 6 Annual abundance of splittail: A) FMWT, all sizes, September-December; B) Bay Study midwater trawl age-0, May-October; C) Bay Study midwater trawl age-1, February-October; D) Bay Study midwater trawl age-2+, February-October.

Striped bass

The striped bass (*Morone saxitilus*) is an introduced anadromous species that supports a valuable sport fishery. Striped bass reproduces in spring in the rivers and rears in fresh and brackish water areas of the estuary. Females mature at age 4 or 5, males at age 2 or 3, with fish living to 20 years. The population of legal-size fish was probably 3 to 4.5 million in the early 1960s, 1.9 million in the early 1970s, 600,000 in 1994, and 1.5 million in 2000, the last year for which a population estimate has been calculated. Based on our understanding of factors controlling striped bass abundance in the estuary (Stevens et al. 1985), this most recent adult population increase was unexpected and remains unexplained. In contrast to the adult population, age-0 striped bass abundance has been low since the mid-1980s, with the lowest indices in the past 3 years. The age-0 striped bass decline is corroborated by all of the IEP long-term monitoring programs.

The 2004 TNS striped bass 38-mm index was 0.8, with a set date of July 13, 2004. This was the lowest index in the 45-year history of the survey, and continues the trend of low indices since 1986 (Figure 7A). Although the previous record low index of 1.4 was in 1998, the 2003 index was 1.5 and the 2002 index could not be set, probably due to extremely low abundance. The 2004 Suisun Index was 0.3 and the Delta Index was 0.5.

Striped bass were found in every area during the 1st TNS survey, with the majority collected in the Sacramento River (Table 2). In the 2nd survey, the majority was collected in the San Joaquin River, but the center of distribution moved back to the Sacramento River during the 3rd survey. In the 4th and 5th surveys, all fish were collected in Montezuma Slough, but catches were very low (Table 1). No striped bass were caught in the last survey.

The 2004 FMWT age-0 striped bass index was approximately half the 2003 index, constituting the lowest index of record (Figure 7B). The 2002 to 2004 indices were the 3 lowest for this survey, which may signal a new trend for age-0 striped bass abundance. Unfortunately, this new trend is one of even lower abundance than the decline noted from the mid-1980s to 2001. Distribution of striped bass in the FMWT was centered in Suisun Bay during September and October, but expanded in both directions in later months. By December, San Pablo Bay accounted for the largest percentage of the index (28%).

Table 2 Mean length, sample size, and survey indices for striped bass and delta smelt for Townet surveys 1-6, 2004.

	Survey 1	Survey 2	Survey 3	Survey 4	Survey 5	Survey 6
Striped Bass						
Mean length (mm FL)	23.8	26.1	38.7	71	63.7	0
N	112	53	24	2	3	0
Survey index	2.3	1.8	0.8	0	0	0
Delta Smelt						
Mean length (mm FL)	34	36	37	44	48	50
N	75	53	16	27	9	8
Survey index	3.6	2.1	0.6	1.0	0.4	0.3

Both the 2004 Bay Study otter trawl and midwater trawl age-0 striped bass abundance indices were the 2nd lowest for the study period (Figures 7C and 7D), only slightly higher than the record low indices of 2002. As for the TNS and FMWT, Bay Study age-0 striped bass indices have been relatively low since the mid-1980s, with the past 3 years at record lows. Overall, the otter trawl collects far more age-0 striped bass than the midwater trawl, and 2004 was not an exception. From May-December, the otter trawl collected 405 age-0 bass while the midwater trawl collected only 46.

Initially the Bay Study collected age-0 striped bass from Suisun Bay through the lower Sacramento and San Joaquin rivers, with the distribution slowly expanding downstream through summer and fall to include Carquinez Strait. Age-0 striped bass were also strongly associated with the shoals, with 91% (n=542) of all fish collected at shoal stations by the otter trawl in 2004.
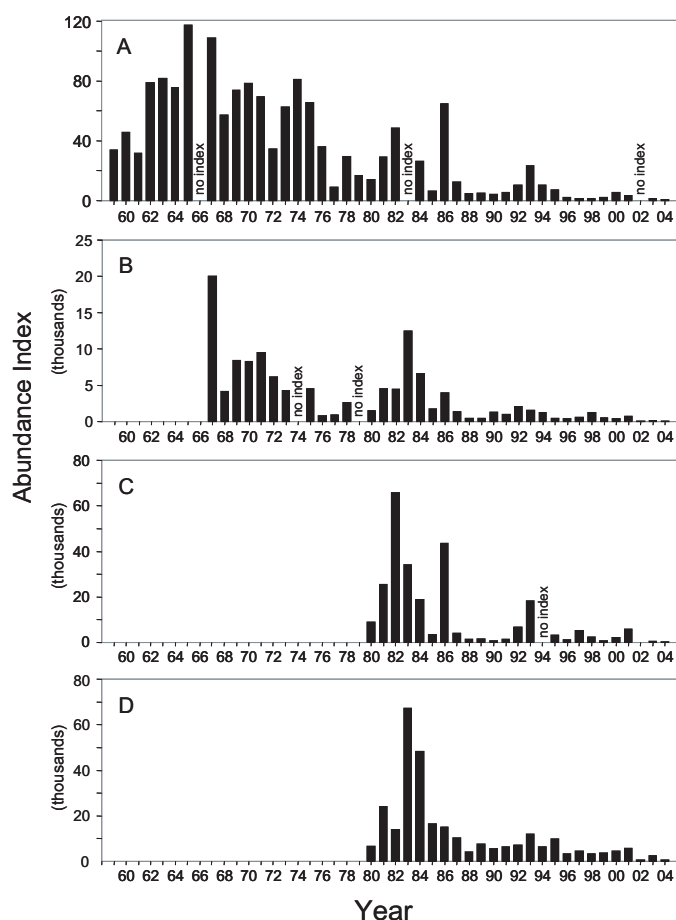

Figure 7 Annual abundance of age-0 striped bass: A. TNS 38-mm index; B. FMWT, September-December; C. Bay Study midwater trawl, June-October; D. Bay Study otter trawl, June-October

Upper Estuary Demersal Fishes

Shokihaze goby

The introduced shokihaze goby, *Tridentiger barbatus,* was first collected in the estuary by the Bay Study in 1997. Since it is common upstream of our original sampling area, abundance is calculated as the annual mean catch-per-unit effort (CPUE, #/hectare) for all 52 stations sampled, including the lower Sacramento and San Joaquin river stations added in 1991 and 1994. In 2004, mean CPUE for fish >19 mm total length (TL) decreased from the previous year, yet remained above 1997-2004 mean of 3.62 fish per hectare (Figure 8). The 2004 shokihaze goby catch exceeded our combined catch of the 2 other introduced *Tridentiger gobies*, the shimofuri goby (*T. bifasciatus*) and the chameleon goby (*T. trigonocephalus*), and was slightly less than the total catch of the yellowfin goby.

In 2004 shokihaze gobies were collected in South Bay at our channel stations north and south of the Dumbarton Bridge and from San Pablo Bay through the western delta. The majority (80%, n=210) of fish was collected from the channel stations in Suisun Bay and the lower Sacramento River.

Yellowfin goby

Another introduced goby, the yellowfin goby (*Acanthogobius flavimanus*), is found throughout the estuary but is most common in shallow brackish and fresh water habitats. The 2004 yellowfin goby age-0 index decreased from 2003 and was only 23% of the 1980-2004 average index (Figure 9). In 2004 yellowfin gobies were collected from South Bay through the western delta, with the majority from Suisun Bay (50%, n=129) and the lower Sacramento and San Joaquin rivers (32%, n=87).
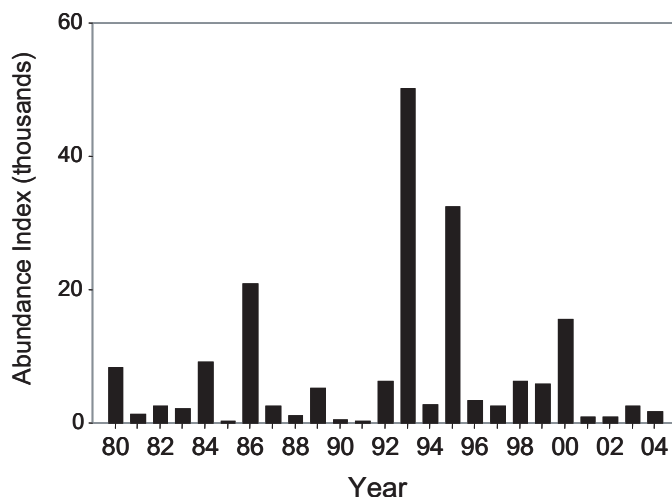

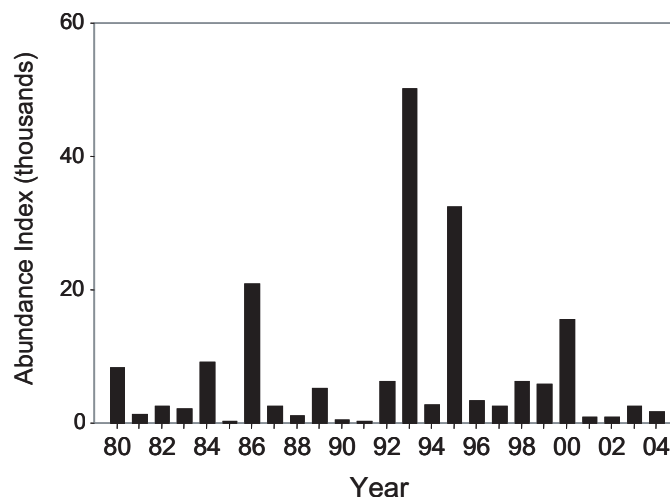Figure 8 Annual CPUE (#/hectare) of shokihaze goby, Bay Study otter trawl, January-December.

Figure 9 Annual abundance of age-0 yellowfin goby, Bay Study otter trawl, May-October.

Starry flounder

The starry flounder (*Platichthys stellatus*) is an estuary-dependent species that spawns in the ocean, but rears in brackish-water areas of estuaries. The 2004 age-0 starry flounder index was about 35% of the 2003 index (Figure 10A), but was well above the very low indices from 1987-1994 and 2000-2002. Starry flounder is also a cold-temperate species and the decline may have been due to the slightly warmer ocean temperatures in winter 2003-04. We collected age-0 starry flounder from June to December from our San Pablo Bay shoal stations to just

upstream of the Rio Vista Bridge on the Sacramento River and to our furthest upstream station on the San Joaquin River, at Old River Flats. Catches were highest at Suisun Bay and Sacramento River shoal stations.

The 2004 age-1 starry flounder abundance index was well below average, and was nearly identical to the 2003 index (Figure 10B). The abundance of age-1 starry flounder has been positively correlated with spring freshwater outflow from the previous year (Kimmerer 2002); since the 2003 March-May outflow was almost double the 2002 March-May outflow, an increase in the 2004 age-1 index was expected. One age-1 starry flounder was collected in South Bay, the remainder from San Pablo Bay upstream to the lower Sacramento and San Joaquin rivers. Distribution was broadest from January through April, when abundance was highest. For the remainder of the year, most fish were collected from San Pablo and Suisun bays. Also, 90% (n=45) of all age-1 starry flounder was collected at shoal stations.
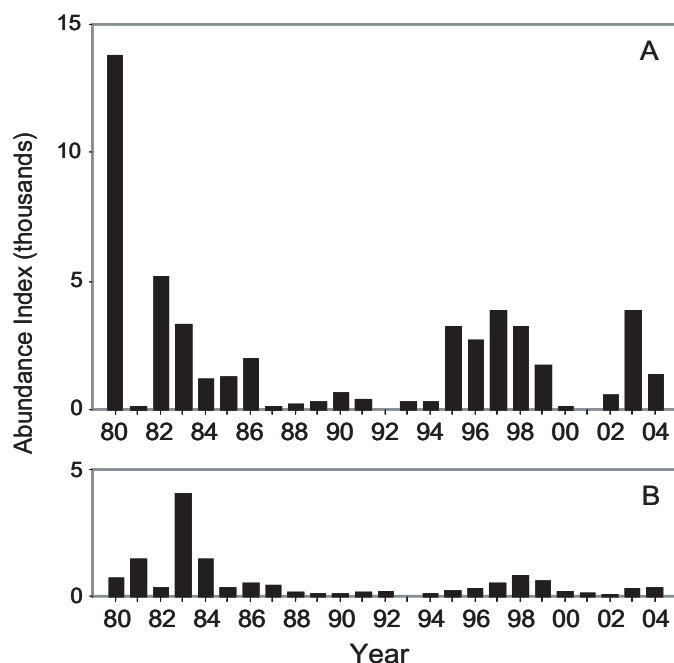

Figure 11 Annual abundance of age-0 Pacific herring, Bay Study midwater trawl, April-September.

Figure 10 Annual abundance of starry flounder, Bay Study otter trawl: A. Age-0, May-October; B. Age-1, February-October

Marine Pelagic Fishes
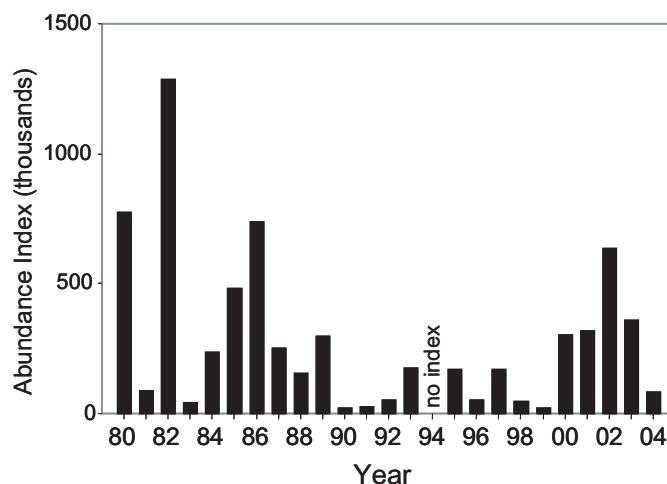
Pacific herring

Pacific herring (*Clupea pallasii*) is an estuary-dependent species that spawns and rears in higher salinity areas (>20 ‰) of the estuary. The 2004 age-0 Pacific herring abundance index decreased substantially from 2003, and was the lowest index in 5 years (Figure 11). After very low indices through the 1990s, there was a modest increase in age-0 indices from 2000-2003. Age-0 Pacific herring were first collected in April in South, Central, and San Pablo bays. In May and June, we collected age-0 herring from South Bay through Suisun and Honker bays, with the largest catches at stations in southwestern San Pablo Bay and northern Central Bay. Between July and September, age-0 fish moved back into Central Bay and by October, most age-0 Pacific herring had emigrated from the estuary.

Northern anchovy

The northern anchovy (*Engraulis mordax*) is the most common fish in the lower estuary and an important prey species for many fishes and seabirds. The 2004 northern anchovy abundance index was the lowest for the study period and continued a 4-year trend of below average indices (Figure 12). San Francisco Estuary is situated between the northern and central anchovy subpopulations and our catches reflect the size and coastal movements of these subpopulations. The most recent abundance

decrease in the estuary may be due to a southward migration of the central subpopulation in response to the cool ocean regime. Although the central subpopulation is the largest and historically the most heavily fished, there are currently no stock assessments, so we cannot confirm subpopulation movements and size. Northern anchovies were larger in 2004 than in recent years, with a May-December mean size of 90.3 mm FL (n=36,709) compared to 75.9 mm in 2002 (n=79,134) and 78.0 mm in 2003 (n=75,907). We collected northern anchovy throughout South, Central, and San Pablo bays and in Suisun Bay to near Port Chicago. Occasional large collections (>1,000 fish) were made in Central Bay from April through October and Central Bay accounted for 71% (n=32,779) of our total 2004 catch.

The jacksmelt (*Atherinopsis californensis*) seasonally migrates from the coast to bays and estuaries to spawn and rear. Age-0 jacksmelt abundance was slightly lower in 2004 than in either 2003 or 2002, but was still above average for the study period (Figure 13). Four years of above average indices reflect a modest recovery from the low indices of 1990-2000. Juvenile jacksmelt rear in shallow (< 2m) areas of South, Central, and San Pablo bays in late spring and summer; after growing to about 50mm FL they begin to migrate to deeper water, where they become vulnerable to our gear. In 2004, 90% (n=418) of age-0 jacksmelt was collected between July and October and by November, most had emigrated to the ocean. We collected age-0 jacksmelt from lower South Bay to upper San Pablo Bay in 2004, with most from mid and northern South Bay.
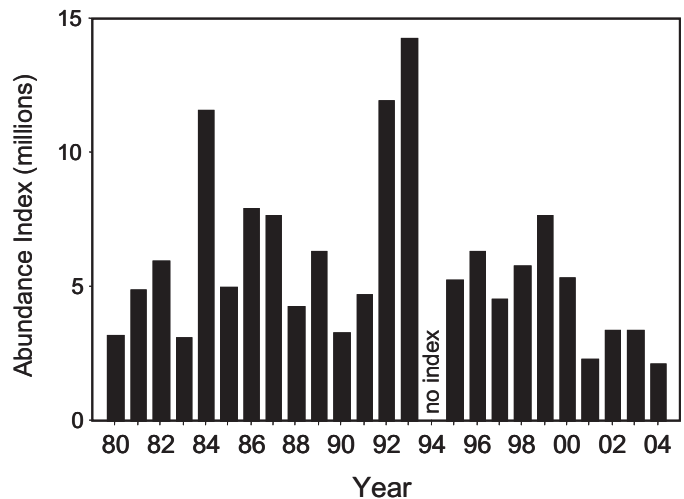

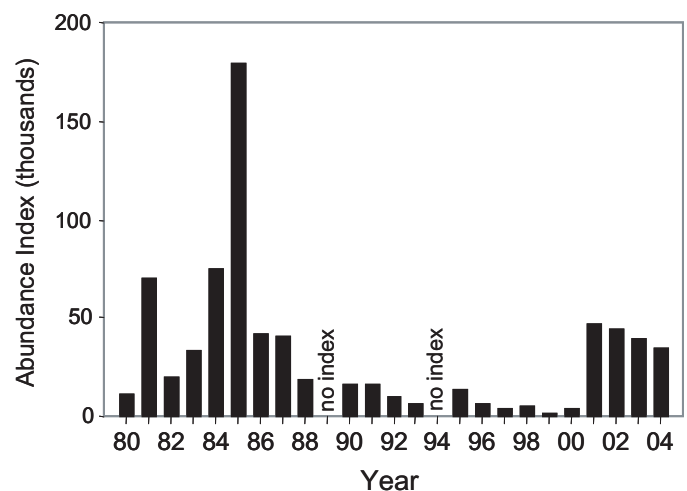**Figure 12 Annual abundance of northern anchovy (all sizes), Bay Study midwater trawl, April-October.**

Figure 13 Annual abundance of age-0 jacksmelt, Bay Study midwater trawl, July-October.

Surfperches

Most of the surfperches are transient species, immigrating to bays and estuaries to give birth to live, fully formed young in late spring and summer. All of the surfperches common to San Francisco Estuary underwent abundance declines in the 1980s per Bay Study trawl and sportfish survey data (DeLeón 1998). Consequently, CDFG changed the sportfish regulations in 2002, adopting a closed season for all surfperches except for shiner perch from April 1 to July 31 in San Francisco Bay. A 5-fish combination bag limit for all species except for shiner

perch and a 20-fish bag limit for shiner perch were also implemented for all areas of California.

In 2004, abundance of age-0 shiner perch (*Cymatogaster aggregata*) increased from 2003 and was the 4th year of a modest recovery from the low indices of the late 1980s and 1990s (Figure 14). Age-0 fish were collected from April through December, with the highest catches in June and July. Fish were initially most common at shoal stations from South through San Pablo bays and migrated through summer to deeper Central Bay stations, resulting in a more even distribution between channel and shoal stations. From April to July 94% (n=442) of all age-0 shiner perch were collected from shoal stations; from August to December 52% (n=139) were collected from Central Bay channel stations

Age-0 walleye surfperch (*Hyperprosopon argenteum*) abundance decreased by approximately 63% in 2004 from 2003, and was only 43% of the study period average (Figure 15). This decrease ended a 3-year period of above average indices. Only 18 age-0 walleye surfperch were collected in 4 tows at 3 stations by the midwater trawl in 2004. In contrast, the age-1+ index increased, following the higher age-0 indices of 2001-2003 (Figure15). We collected 32 age-1+ fish in 16 tows at 9 stations. All but 1 of the 50 total walleye surfperch collected were from Central and San Pablo bays and all but 1 were from shoal stations. The highest catches were from our station near the Berkeley Fishing Pier and the 2 stations nearest to Point Pinole; these 3 stations combined accounted for 84% (n=42) of the total catch.

The 2004 white seaperch (*Phanerodon furcatus*) abundance index increased for the 4th consecutive year and was the highest since 1984 (Table 3). We collected a total of 37 white surfperch in 2004, but only 9 were from stations and months used for index calculation. All white seaperch were collected from Central Bay and all but 2 were from shoal stations.
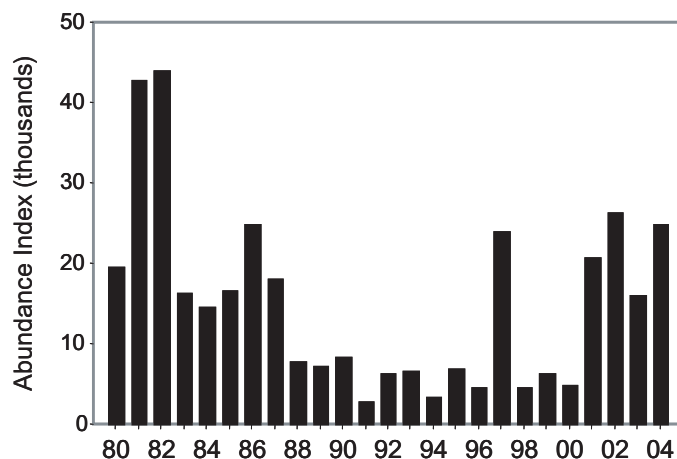

Figure 14 Annual abundance of age-0 shiner perch, Bay Study otter trawl, May-October.
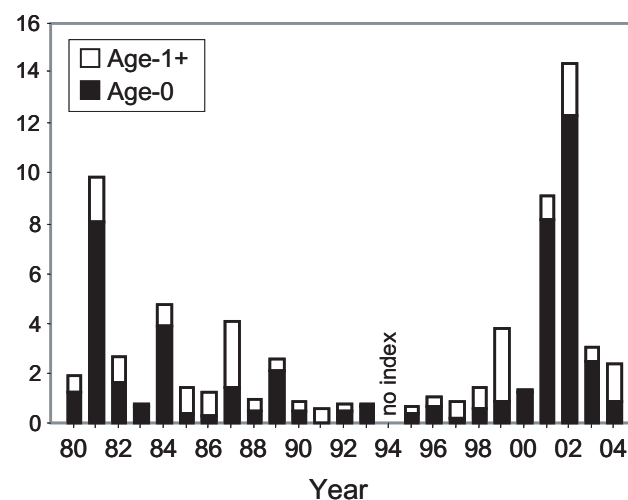

Figure 15. Annual abundance of age-0 and age-1+ walleye surfperch, Bay Study midwater trawl, May-October.

Table 3 Annual Bay Study abundance indices for selected surfperch species from the otter trawl. The white seaperch (all sizes) and pile perch age-0 indices are from May-October while the barred perch (all sizes) index is from April-September.

Year	white seaperch	barred surfperch	pile perch
	all	all	age-0
1980	588	455	857
1981	1248	942	998
1982	349	335	471
1983	271	1330	778
1984	873	673	110
1985	138	73	301
1986	309	0	254
1987	265	239	0
1988	148	134	0
1989	48	101	153
1990	95	79	0
1991	0	84	0
1992	0	41	0
1993	0	43	0
1994	0	80	0
1995	0	0	0
1996	0	59	0
1997	0	155	0
1998	36	48	75
1999	0	46	0
2000	0	43	31
2001	106	55	0
2002	260	59	42
2003	371	352	0
2004	487	115	0

The barred surfperch (*Amphistichus argenteus*) abundance index decreased by 67% from 2003 to 2004, but was still the 3rd highest index since 1988 (Table 3). Only 2 barred surfperch were collected during 2004; both were from South Bay shoal stations.

The 2004 pile perch (*Rhacochilus vacca*) age-0 abundance index was again 0, showing no sign of recovery in the estuary and continued the trend of very low or 0 indices since 1987 (Table 3). This is the most dramatic decline of all the surfperches, as pile perch was the 3rd most common surfperch collected by the Bay Study in the early 1980s. One pile perch was collected in March 2004,

prior to the index period, at the shoal station near Treasure Island.

Several surfperches, including shiner perch, white seaperch, and possibly walleye surfperch, may be recovering from the low population levels of the late 1980s and 1990s. However, our annual catches for all species, except for shiner perch, are still so low and sporadic that we can not yet concluded that there has been a definite recovery or if sportfish regulations implemented in 2002 have had a quantifiable benefit.

Marine Demersal Fishes

Brown smoothhound

The brown smoothhound (*Mustelus henlei*) is the most common shark collected by the Bay Study in the estuary. It immigrates to bays and estuaries to pup in late spring and summer and young fish emigrate to the ocean in fall. Although the 2004 age-0 brown smoothhound abundance index was more than double the 2003 index, it was only 71% of study period average (Figure 16). We collected age-0 brown smoothhound from June through October, with 85% (n=23) from June through August. Most fish (78%, n=21) were collected at channels stations throughout South, Central, and San Pablo bays.
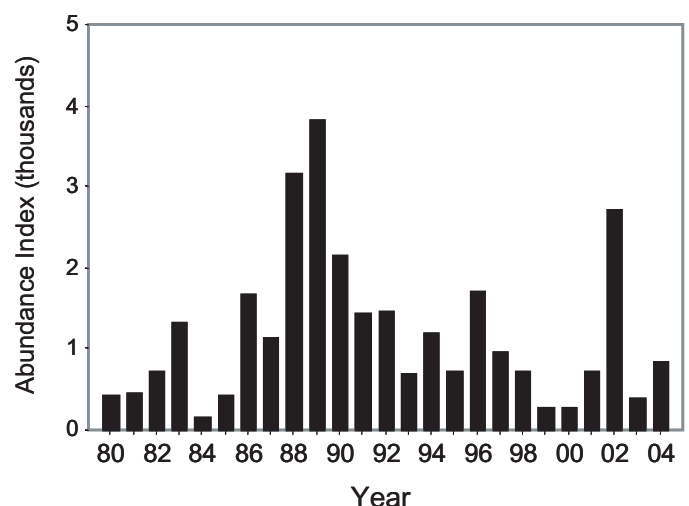

Figure 16. Annual abundance of age-0 brown smoothhound, Bay Study otter trawl, April-October.
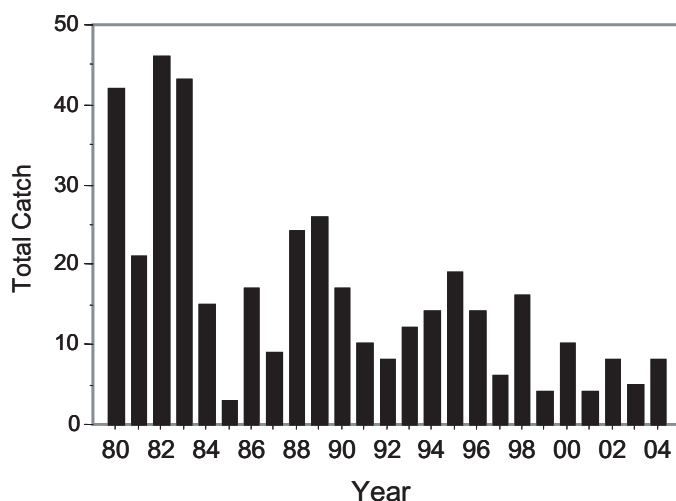
Figure 17 Annual catch leopard shark (all sizes), Bay Study otter trawl, all months.
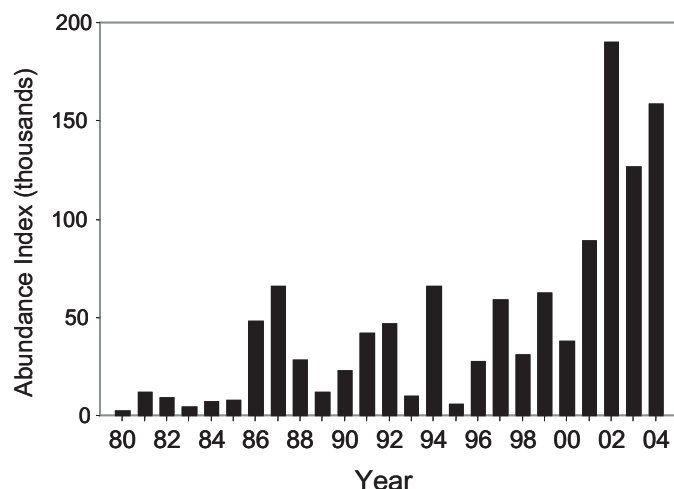

Figure 18. Annual abundance of age-0 plainfin midshipman, Bay Study otter trawl, June-October.

Leopard shark

The leopard shark (*Triakis semifasciata*), is a popular sportfish that immigrates to very shallow areas of the estuary, especially in South Bay, to pup in summer. The Bay Study does not effectively sample age-0 leopard sharks because they are born and rear in areas too shallow to sample with our boat and the otter trawl. Because catches are often very low, we report total catch rather than abundance indices. Our 2004 otter trawl catch (all ages) increased slightly from 2003 (Figure 17). There has been a downward trend in catch since 1984, with an apparent step change in 1999. Catch averaged 38 fish per year from 1980 to 1983, declined to 14 fish per year from 1984 to

1998, and declined again to only 7 fish per year from 1999 to 2004. Because of potential over harvest of leopard sharks, a 36-inch size limit and a 3-fish bag limit were implemented in 1991 for the sport fishery.

All 15 leopard sharks collected in 2004 were from stations south of the Bay Bridge. During the study period, nearly 85% (n=468) of leopard sharks were collected from stations south of the Bay Bridge and approximately 10% (n=57) and 5% (n=31) were collected from Central Bay and San Pablo Bay, respectively. Also, 80% (n=460) were collected from channel stations. Most of the leopard sharks we collect are migrating to or from the South Bay shoals.

Plainfin midshipman

The plainfin midshipman (*Porichthys notatus*) migrates from coastal areas to bays and estuaries in late spring and summer to spawn. Most juveniles rear in the estuary though December, with some fish remaining through winter. The 2004 age-0 index was the 2nd highest for the study period and was the 4th consecutive year of record high indices (Figure 18). Age-0 plainfin midshipmen were first collected in June and were most common in September. Distribution was broadest in August, with fish collected from South Bay near the San Mateo Bridge to near Chipps Island. The majority of age-0 plainfin midshipman (58%, n=2,024) was collected from the Central Bay channel stations in 2004.

Pacific staghorn sculpin

The Pacific staghorn sculpin (*Leptocottus armatus*) is a common species that usually rears in higher salinity areas, but is not uncommon in brackish water and is occasionally found in fresh water. Throughout the estuary it rears in intertidal and shallow subtidal areas from late-winter through early summer. The 2004 Pacific staghorn sculpin age-0 abundance index was 48% of the 2003 index, and was the lowest since 2000 (Figure 19). However, record high indices occurred from 2001 to 2003 and the 2004 index was just below the study period average. Age-0 fish were first collected in February and March in San Pablo and Suisun Bays. Distribution broadened in April and May, with fish collected from South Bay to the lower Sacramento and San Joaquin rivers. Migration of age-0 fish to Central Bay began in June and continued through September. No age-0 fish were collected after September, as most had emigrated from the estuary.
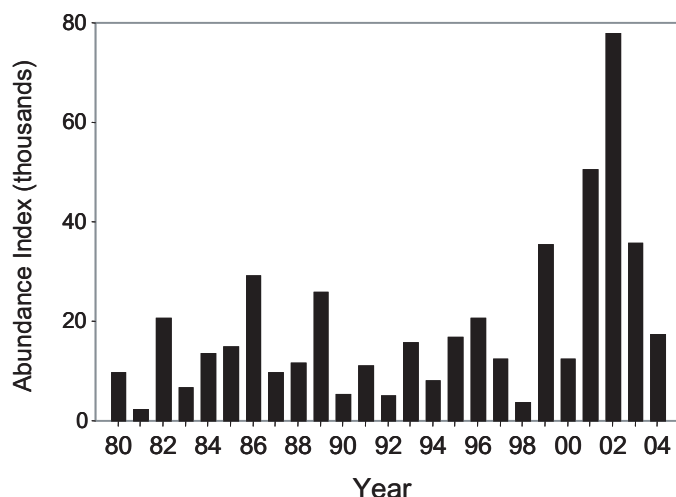
Figure 19 Annual abundance of age-0 staghorn sculpin, Bay Study otter trawl, February-September.
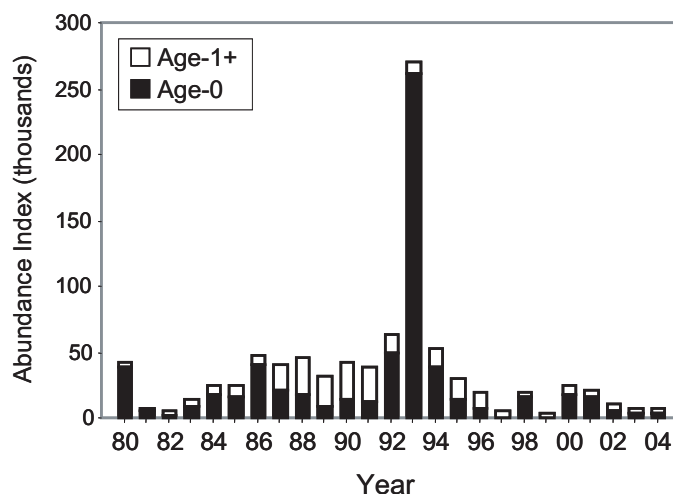

Figure 20 Annual abundance of age-0 and age-1+ white croaker, Bay Study otter trawl, February-October

White croaker

The white croaker (*Genyonemus lineatus*) is a common coastal species that frequents bays and estuaries. The 2004 age-0 white croaker abundance index was nearly identical to the 2003 index and the 5th lowest for the study period, about 15% of average (Figure 20). Age-0 catch has been below average for the past 10 years and exceptionally low the past 3 years. The age-1+ index increased, but was only about 33% of average (Figure 20). It was the 3rd lowest index since 1982 and the 8th consecutive year of below average indices. White croaker is a warm-sub-tropical marine species and as such, age-0 abundance in San Francisco Estuary was related positively to elevated ocean temperatures. Age-1+ abundance was highest during the 1987-1992 drought, when salinities were high and relatively stable year-round in the estuary. Age-0 white croaker were collected from April-December, whereas age-1+ were collected every month except November. In 2004 white croaker were collected from South Bay through San Pablo Bay and had the widest distribution from January through May; by summer the majority had migrated to Central Bay and by November most had returned to the ocean. Overall, 45% (n=120) of the 2004 catch was from our channel station near Angel Island; most of these fish were collected in September and October, just prior to emigration.

Bay goby

The bay goby (*Lepidogobius lepidus*) is one of the most common native gobies in the estuary. It is a resident species that rears in the higher salinity areas and lives no longer than 2-3 years. In 2004, the bay goby abundance index decreased to 27% of the 2003 index, but 2003 was the 2nd highest of the study period (Figure 21). Three of the 4 highest indices occurred from 2001 to 2003, but the 2004 index was below the study period average. From January to May, bay gobies were distributed from South Bay through San Pablo Bay, with a single fish collected in Carquinez Strait, yet the majority (83%, n=2,686) was from Central Bay. From June to October, as water temperatures increased, fish moved from shoals to channels and to Central Bay. In November and December, bay goby catches declined, but the distribution again extended to San Pablo and South bays.
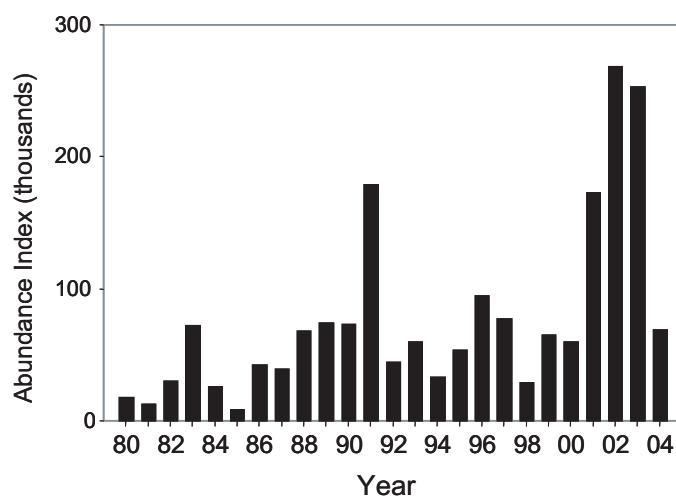
Figure 21 Annual abundance of bay goby (all sizes), Bay Study otter trawl, February-October.

Figure 22 Annual abundance of juvenile (age-0 and age-1) and age-2+ California halibut, Bay Study otter trawl, February-October.

California halibut

The California halibut (*Paralichthys californicus*) is a subtropical species that became common in San Francisco Estuary in the 1980s and 1990s, concurrent with the recent warm-water regime. In 2004, we collected a few age-0 fish after 4 consecutive years of zero indices (Figure 22). The appearance of age-0 fish is believed to be in response to Gulf of the Farallones SSTs reaching 14 °C for some time in 2003. Laboratory experiments have shown high larval mortality at 12 °C and increased survivorship and growth with higher temperatures (Gadomski and Caddell 1991). SSTs exceeded 14 °C in August and September 2003, and we collected age-0 fish in January, April, and November 2004. Five of the 6 age-0 fish were collected at a shoal station south of the San Mateo Bridge in South Bay and 1 was collected near the Mothball Fleet. The age-2+ California halibut index in 2004 decreased from 2003, the 5th consecutive year of decline (Figure 22). Age-2+ fish were collected from South through San Pablo bays; these fish ranged from 330 to 830 mm TL, indicating several year classes are still present in the estuary.
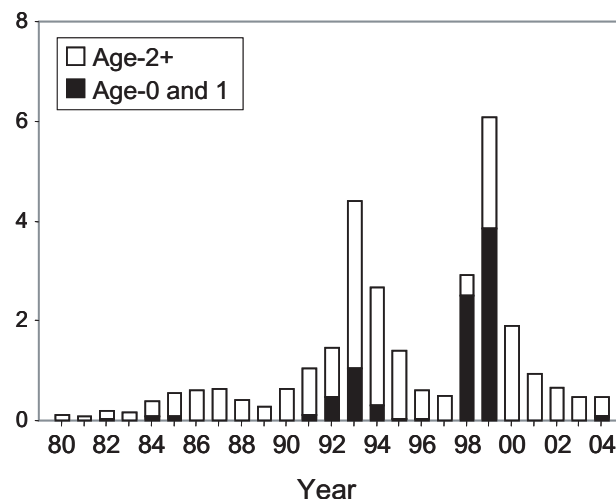
English sole

English sole (*Pleuronectes vetulus*) is a common coastal species that spawns in winter and rears in both the ocean and estuaries. The 2004 age-0 English sole abundance index decreased somewhat from 2003 (Figure 23), but continued the trend of above average indices since 1999. This high abundance corresponds with a shift to a cool-water ocean regime; we believe that English sole, a cold-temperate species, has benefited from these ocean conditions. In several years with strong year classes, including 2000, 2002, and 2003, 2 cohorts have entered the estuary, with the 1st cohort arriving as early as December or January and the 2nd cohort in April or May. However, in 2004 only a single cohort of age-0 English sole entered the estuary; with most fish immigrating in May. Distribution of age-0 English sole in 2004 was typical - fish reared at the shoals from South Bay to Carquinez Strait in spring and early summer, migrated back to the channels and Central Bay through summer, as temperatures at the shoals increased, and emigrated to the ocean in the fall.
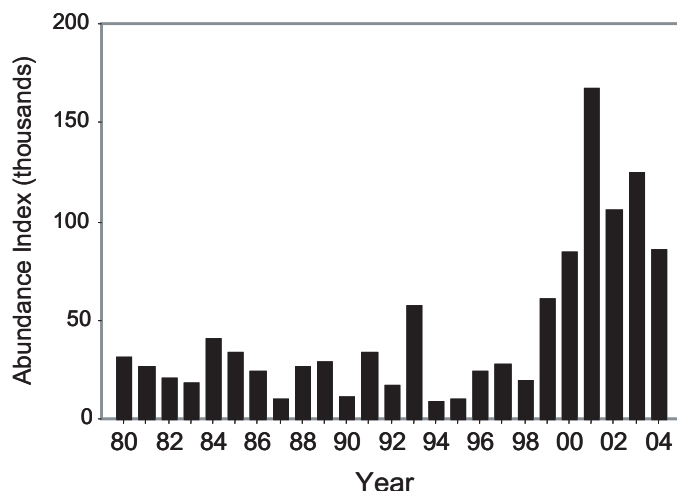
Figure 23 Annual abundance of age-0 English sole, Bay Study otter trawl, February-October.
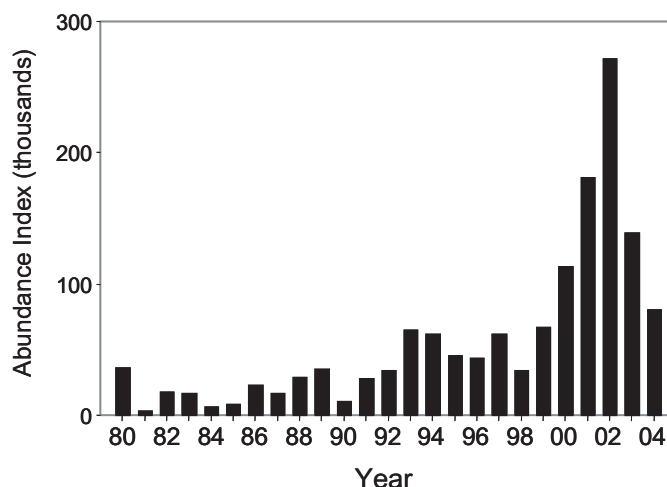

Figure 24 Annual abundance of speckled sanddab (all sizes), Bay Study otter trawl, February-October.

Speckled sanddab

The speckled sanddab (*Citharichthys stigmaeus*) is one of the most abundant flatfishes in the estuary. It spawns along the coast and juveniles migrate into the estuary to rear for up to a year. The 2004 speckled sanddab abundance index decreased to 60% of the 2003 index, yet was the 5th highest index of the study period (Figure 24). Record speckled sanddab abundance indices occurred from 2000 to 2004, corresponding with cooler ocean temperatures and strong summer upwelling. Such conditions could benefit speckled sanddabs, as they commonly spawn in summer, have a very long pelagic period, and do not settle until after the upwelling season ends. Speckled sanddabs were distributed from South through San Pablo bays in 2004, with a single fish collected from Carquinez Strait. Fish in South and San Pablo bays were most common at the shoals, but only from January to May. By June, the majority of speckled sanddabs moved from South and San Pablo bays to Central Bay and remained common there through December.

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Central Valley Chinook Salmon Catch and Escapement

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In 2004, the ocean catch of Central Valley Chinook salmon south of Point Arena increased in both the commercial and recreational fisheries from 2003. However, the catch per unit effort (CPUE) decreased between 2003 and 2004. Compared to the 1970-2004 period of record both the ocean catch and catch per unit effort were above average.

The total escapement of fall, spring, and winter run Chinook salmon decreased in 2004 but remains above the average escapement for the 1970-2004 period of record. In 2004 the fall run Chinook escapement to the Sacramento River system was the lowest since 1998 but remained the greatest contributor to the Central Valley fall run escapement. Spring run escapement to both Mill and Deer creeks decreased from 2003 to 2004 while escapement to Butte Creek increased. Winter run escapement decreased from 2003 to 2004 but was the third highest escapement since 1981. The three-year cohort replacement rate also indicates that population is continuing the upward trend started in 1995.

Central Valley Chinook Fall Run Ocean Harvest Index and Ocean Catch

The Pacific Fisheries Management Council (PFMC) sets a spawner escapement goal for Sacramento River system fall run Chinook. They also develop harvest regulations to protect listed Central Valley winter and spring-run Chinook as well as Klamath River fall run Chinook. These include setting minimum size limits, gear restrictions and season restrictions south of Point Arena. These regulations restrict harvest of all Chinook runs.

The PFMC's Central Valley Chinook ocean harvest index (OHI) is an approximate harvest rate. The OHI is calculated by dividing the total ocean catch south of Point Arena by the catch plus escapement. The ocean harvest index does not include inland harvest, which may account for up to 25% of the returning adults. In 2004, the OHI increased to 62% due to the increased ocean harvest (Figure 1). The Central Valley Chinook escapement decreased to 334,300 spawners (Figure 1).

Statewide the ocean catch increased between 2003 and 2004. For the commercial fishery, the number of days fished (boat days) increased from 15,900 in 2003 to 21,600 in 2004 but the CPUE (estimated total number of fish caught / total number of boat days fished) decreased from 30.9 fish/day to 23.2 fish/day (Figure 2). The CPUE also decreased in Washington and Oregon but remained well above average for the 1970-2004 period (Figure 2).
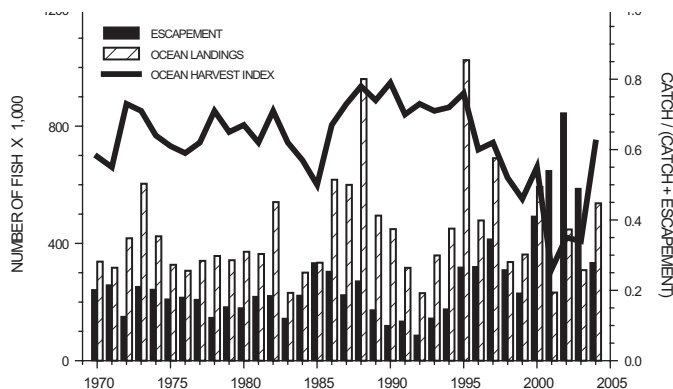
Figure 1 PFMC Chinook salmon ocean catch, the Central Valley fall-run Chinook total adult spawner escapement and ocean harvest index, 1970-2004.
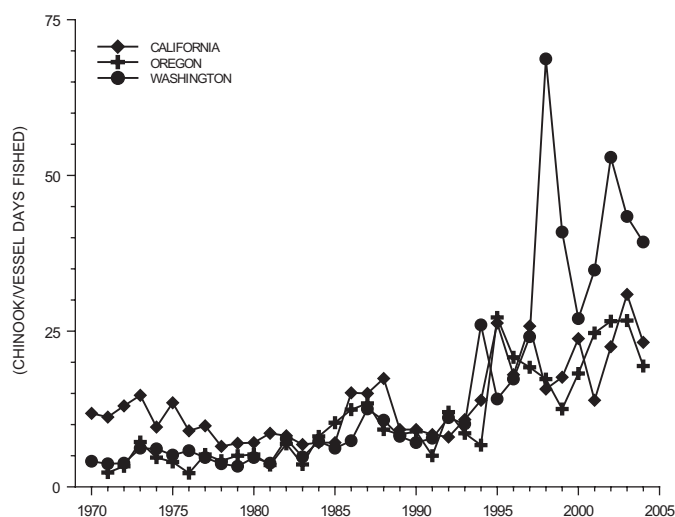

Figure 2 Chinook salmon catch per unit effort (estimated total number of fish caught / total number of boat days fished) in the California, Oregon, and Washington commercial trolls, 1970-2004.

Central Valley Fall run Chinook Escapement

Escapement data reported to the PFMC are partitioned into "natural" and "hatchery" categories. Natural escapement includes all fish returning to spawn in natural areas; these fish are of both natural and hatchery origin. Available data indicate that hatchery-produced fish constitute a majority of the natural fall run Chinook spawners in the Central Valley. Hatchery escapement includes all fish returning to the hatcheries; these fish are also of both natural and hatchery origin. These terms, as defined here, are used throughout this paper and in each of the figures.

The fall run Chinook escapement to the mainstem Sacramento River, the Feather River, the American River, and the Yuba River decreased from 2003 levels but were still higher than the average escapement for the 1970-2004 period. In the San Joaquin River system, the fall run Chinook escapement also decreased from 2003 levels but remained slightly above the average escapement for the 1970-2004 period.

In 2004, a spawner escapement goal of 122,000 to 180,000 Sacramento River system fall-run Chinook (hatchery and natural adults combined) guided PFMC management for this stock. The estimated number of natural spawners was 237,000 exceeding the PFMC management goals (Figure 3). I calculated the cohort escapement by adding three-year olds from the current year and two-year olds from the previous year. The cohort escapement decreased to the fourteenth highest level in the last three decades at about 223,000 (Figure 4).
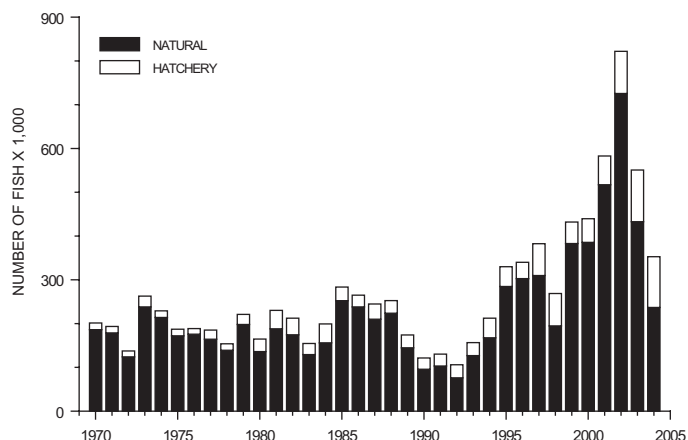

Figure 3 Annual fall-run escapement to the Sacramento River and major tributaries, natural and hatchery, 1970-2004.
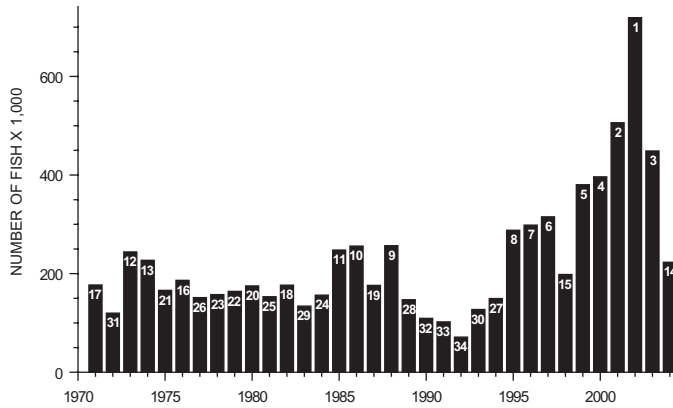
Figure 4 Annual natural fall-run cohort escapement to the Sacramento River and major tributaries, 1970-2004.

Natural spawner escapement to the mainstem Sacramento River decreased from 2003 levels and dropped below the average escapement for the 1970-2004 period for the first time since 1998 (Figure 5). Natural spawner escapement in the American River decreased from about 146,000 in 2003 to 96,000 in 2004 but remained above the average escapement for the 1970-2004 period (Figure 6). In the Feather River, the estimated escapement decreased from 89,000 in 2003 to 48,000 in 2004 but also remained above the average escapement for the 1970-2004 period (Figure 7). The estimated Yuba River fall run escapement decreased from 29,000 in 2003 to 15,000 in 2004 and dropped below the average escapement for the 1970-2004 period (Figure 8).

Figure 5 Annual fall-run escapement to the mainstem Sacramento River, natural and hatchery contribution, 1970-2004.

Figure 6 Annual fall-run escapement to the American River, natural and hatchery contribution, 1970-2004.

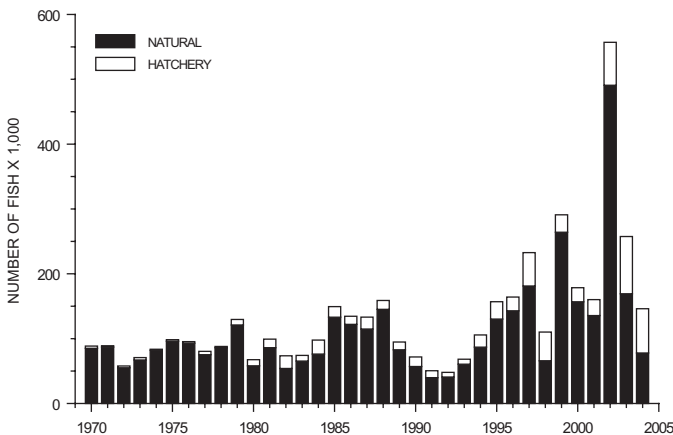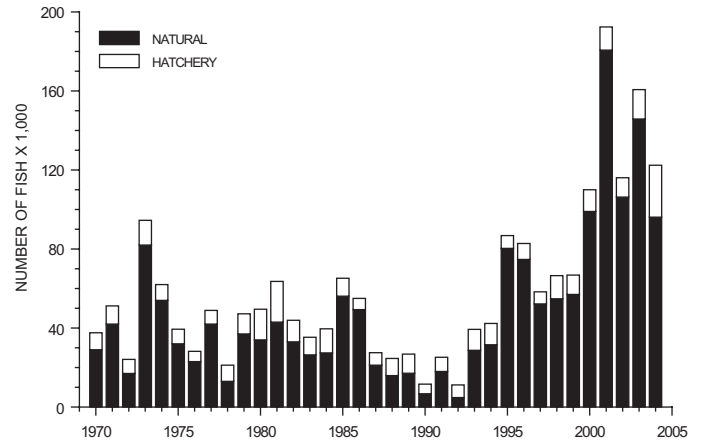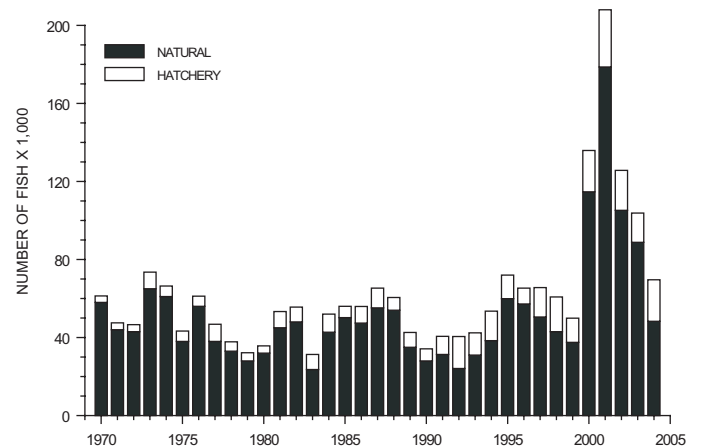**Figure 7 Annual fall-run escapement to the Feather River, natural and hatchery contribution, 1970-2004.**
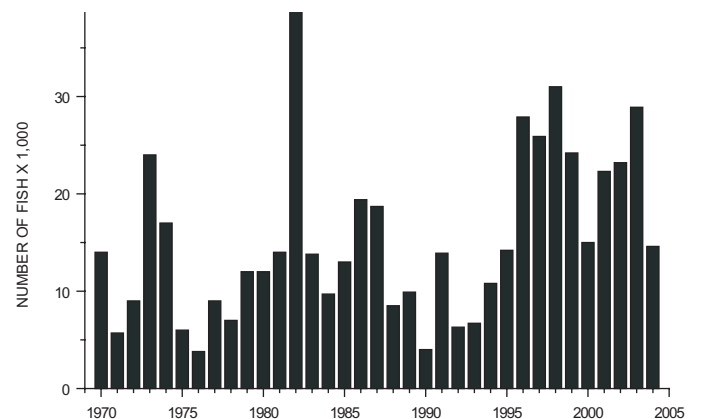

Figure 8 Annual natural fall-run escapement to the Yuba River, 1970-2004.
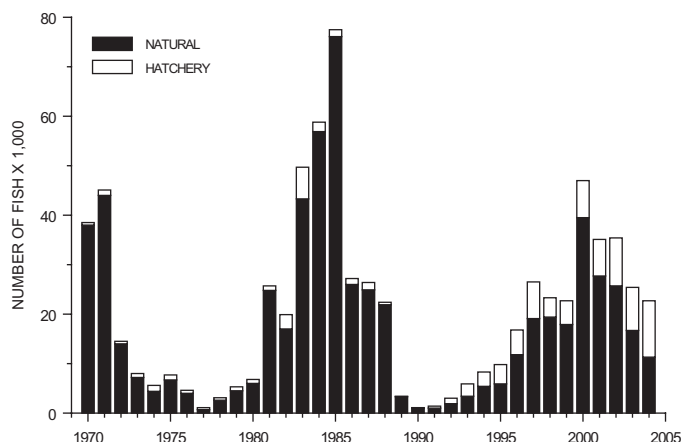
Figure 9 Annual fall run escapement to the San Joaquin River system, natural and hatchery contribution, 1970-2004.
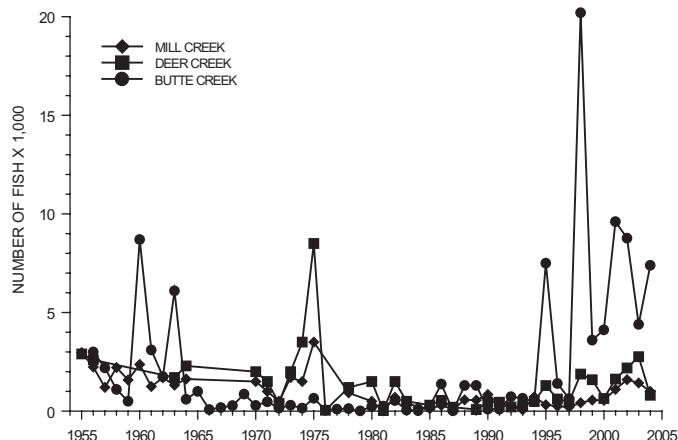

Figure 10 Annual spring run escapement to Mill, Deer, and Butte creeks, 1956-2004.

On the San Joaquin River system, the estimated natural spawner escapement decreased to about 11,000 in 2004 which is the lowest level since 1996 (Figure 9). The escapement was less than half the escapement from three-years earlier and dropped below the average escapement for the 1970-2004 period. In 2004, the hatchery spawners accounted for approximately 50% of the total escapement which is more than double the average number of hatchery spawners for the 1970-2004 period (Figure 9). The San Joaquin River system includes spawners from the Mokelumne, Stanislaus, Tuolumne, and Merced rivers and has constituted less than 10% of the total Central Valley spawner escapement since 1986.

Sacramento River System Spring Run Chinook Escapement

In 2004 the escapement to Deer Creek decreased to approximately 840 natural spawners (Figure 10). The number of spawners was substantially lower than the estimated 1620 spawners from three years earlier (Figure 10). The number of natural spawners also decreased on Mill Creek with an estimated escapement of 1000 which was slightly under the estimated escapement of 1100 three years earlier (Figure 10).

The Butte Creek escapement increased from about 4,400 in 2003 to 7,400 in 2004 based upon a snorkel survey methodology (Figure 10). However, the estimate escapement decreased from the estimated 9,600 spawners three years earlier. DFG has also been using carcass surveys to estimate escapement on Butte Creek since 2001. Based on results from the carcass survey, the estimated escapement of natural spawners decreased from 11,200 in 2003 to 10,200 in 2004. The estimated escapement to Butte Creek continues to surpass the other spring run tributaries and the mainstem Sacramento River (Figure 10).

Winter Run Escapement to the Sacramento River below Keswick Dam

The estimated in-river escapement of winter run Chinook in 2004 was about 8,600 fish based on mark-recapture carcass survey data. This was the highest total escapement estimated since 1981. However, there were a relatively high proportion of two-year-old males in the 2004 spawning returns. The number of adult females returning to spawn in 2004 was lower than in the three previous years.

Escapement estimates based on extrapolated counts at Red Bluff Diversion Dam from 1967 through 2004 were examined for long-term population trends (Figure 11). A cohort replacement rate was calculated by dividing the sum of the current year's three-year olds and the previous year's two-year olds by the same value from three years

earlier. This cohort replacement rate was 2.5 in 2004 based on Red Bluff Diversion Dam data (Figure 11).
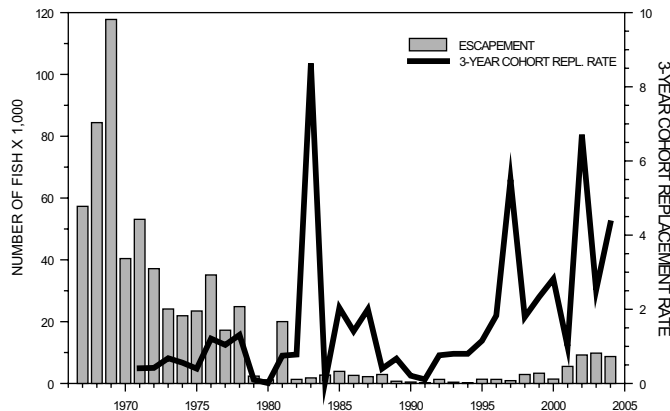
Figure 11. Annual winter-run escapement to the upper Sacramento River and the three-year cohort replacement rate based on extrapolated counts at Red Bluff Diversion Dam,1967-2004.

Most of the data presented in this article is published in the PFMC's Review of the 2004 Ocean Salmon Fisheries report. A copy of the report is available by calling (503) 820-2280 or online at www.pcouncil.org. I thank Colleen Harvey Arrison (DFG) for providing the spring run Chinook escapement data for Mill and Deer creeks

and Tracy McReynolds (DFG) for providing the spring run Chinook escapement data for Butte Creek.

Delta Water Project Operations

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October 2004 to Mid-March 2005 Conditions

During October 2004 to mid-March 2005, daily Sacramento River flows ranged between 250 and 1700 cubic meters per second (8,830 cfs and 60,000 cfs) as shown in Figure 1, with the largest peak of about 1,600 cubic meters per second (56,500 cfs) on January 2, 2005. San Joaquin River flow ranged between 30 and 300 cubic meters per second (1,060 cfs and 10,590 cfs), with the largest peak of 242 cubic meters per second (8,545 cfs) on January 13, 2005. Net Delta Outflow Index (NDOI) ranged between 70 and 1800 cubic meters per second (2,470 cfs and 63,560 cfs), with the largest outflow index of about 1,730 cubic meters per second (61,087 cfs) on January 2, 2005.
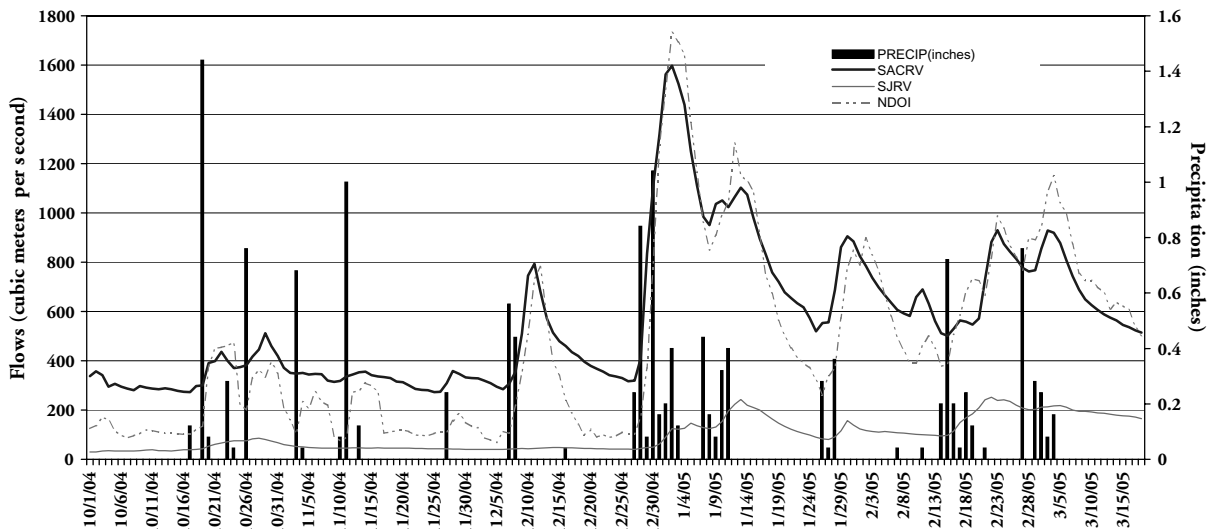
Figure 1   Sacramento River, San Joaquin River, Net Delta Outflow, and Precipitation, October 2004 through mid-March 2005.

During October 2004 to mid-March 2005, daily export actions at the State Water Project (SWP) ranged between 25 and 250 cubic meters per second (882 cfs and 8,827 cfs), and at the Central Valley Project (CVP) it ranged between 45 and 150 cubic meters per second (1,588 cfs and 5,297 cfs). CVP export action was more stable than SWP; as shown in Figure 2. Typically, CVP pumped between 100 and 150 cubic meters per second (3,530 cfs and 5,296 cfs), except on two occasions. The decreased pumping at CVP around December 7, 2004 was for water quality concern, whereas at the end of January/ early February, CVP export reductions were to meet EI ratio standard and fishery action (i.e. Delta smelt), respectively. SWP low pumping during the following periods were due to the following reasons:

- Early half of October 2004 and the latter half of November 2004 were to meet outflow standard.

- Mid-December 2004 and end of December 2004 were to meet water quality concerns

- Early February 2005 to meet EI ratio standard and also for fishery concerns (i.e. Delta smelt)

During the October 2004 to mid-March 2005 period, the first onset of precipitation occurred in mid-October 2004 as shown in Figure 1. The largest daily precipitation event during this period occurred on October 19 with a daily total of 1.44 inches. Thereafter, precipitation continues to be sporadic throughout the period with the most precipitation events occurring in December 2004 for a monthly total of 3.4 inches. As a result, Net Delta Outflow Index was the highest in December too.

Percent inflow diverted during October 2004 to mid-March 2005 was met as shown in Figure 3. From October 2004 through January 2005, the standard is 65% using the 3-day running average, and from February to mid-March of 2005 the standard is 35% using the 14-day running average.

Figure 2 State Water Project and Central Valley Project Pumping, October 2004 through mid-March 2005

Figure 3 Percent Inflow Diverted, October 2003 through mid-March 2005

CONTRIBUTED PAPERS

Salvage of Hatchery-Released Juvenile Steelhead at the State Water Project and Central Valley Project Fish Facilities.

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Introduction

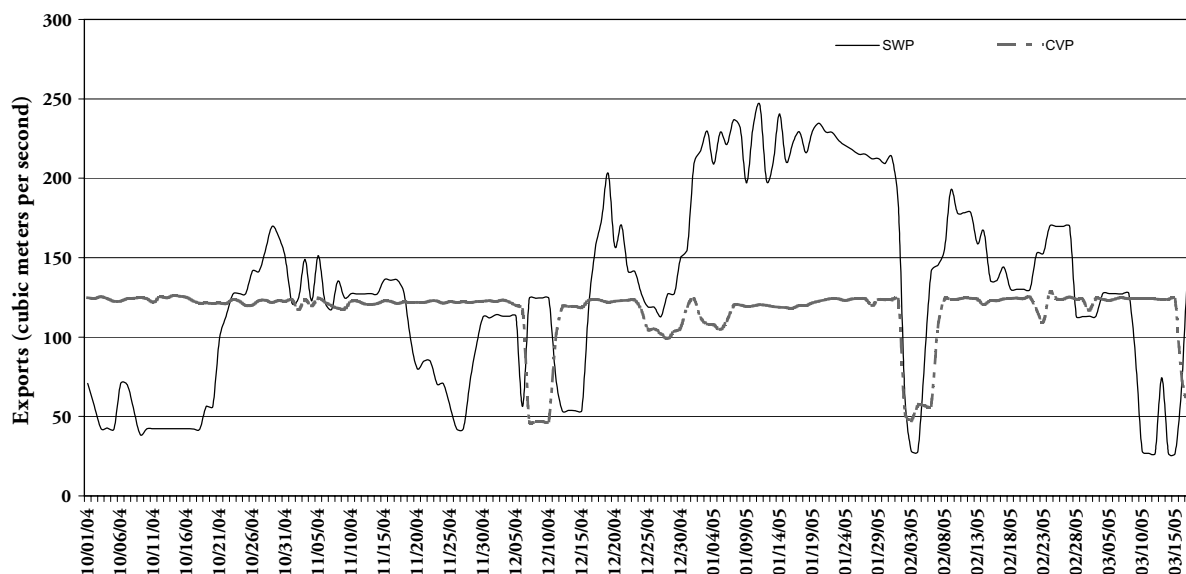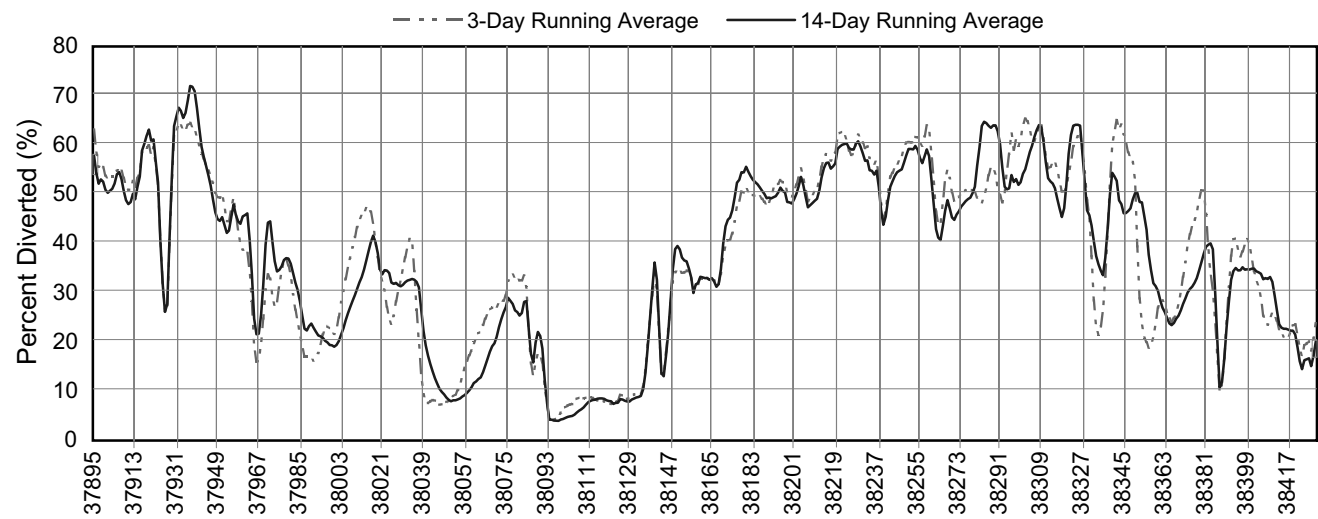Four Central Valley anadromous fish hatcheries release Central Valley steelhead into the Sacramento and San Joaquin rivers or their tributaries (McEwan 2001). Coleman National Fish Hatchery (a federal hatchery located on Battle Creek in the upper Sacramento River), Nimbus Hatchery (located on the American River), and the Feather River Hatchery all release steelhead yearlings directly into the Sacramento River or its tributaries. The Mokelumne River Fish Installation releases steelhead into the Mokelumne River, a tributary of the San Joaquin River that flows directly into the interior Delta.

Two large fish salvage facilities in the southern Delta of the San Francisco Bay Estuary, the Central Valley Project's (CVP) Tracy Fish Collection Facility and the State Water Project's (SWP) Skinner Delta Fish Protective Facility, divert (salvage) fish from exported water. Both facilities use a louver-bypass system to collect fish, which are then transported to release sites in the Delta. Although the salvage process does reduce the losses of steelhead, SWP and CVP water diversions from the Sacramento-San Joaquin Delta-estuary adversely affect steelhead through increased exposure to predators, disruption of their migration route, and through direct mortality from salvage operations. To date, no studies on the effect of the Delta water operations on emigrating hatchery-released steelhead have been done.

Mokelumne River hatchery releases accounted for about 7% (0-10% annually) of hatchery steelhead released in the Central Valley between 1998 and 2003 (DFG unpublished data). A relatively high percentage of these releases are expected to be salvaged at the south Delta export facilities due to the proximity of the release site to those facilities. Delta hydrology also makes fish released in the Mokelumne River more vulnerable to salvage at the facilities, since a relatively high proportion of the San Joaquin tributary flows are entrained by the export facilities compared to Sacramento River flows. Steelhead from all the other hatcheries are released either directly into the Sacramento River or into Sacramento River tributaries far upstream of the Delta. For these fish to make their way to the export facilities, they must enter the interior Delta through either the Delta Cross Channel, Georgiana Slough or near the confluence of the Sacramento and San Joaquin rivers.

The analysis reported here has two objectives: 1) determine which hatcheries are contributing most to the salvage of steelhead at the water export facilities; and 2) calculate transit times from release to salvage at fish facilities.

Methods

The analysis presented here is based on the source identification of salvaged steelhead with coded wire tags (CWT) applied at the various hatcheries from 2000 to 2004. Salvaged steelhead were examined for adipose fin clips and scanned by a metal-detecting wand for the presence of CWTs. If CWTs were detected, fish were sacrificed for later tag removal. Criteria for sacrifice of CWT steelhead varied. In 2000, all fish with a CWT were sacrificed, but after 2000, only fish 300 mm or smaller were sacrificed.

CWTs from hatchery-released steelhead were extracted and their codes recorded. Release dates for groups of CWT steelhead were obtained from the Regional Mark Information System (RMIS) website. Transit times were calculated by subtracting the date of salvage from the date of release.

From 2000 to 2003, only two hatcheries used CWTs to mark hatchery fish, Coleman National Fish Hatchery and Feather River Hatchery. In 2004, Mokelumne Fish Installation applied CWTs (as did Coleman), but Feather

River Hatchery did not tag any steelhead with CWTs, apparently due to funding restrictions (Bob Kano, personal communication). No steelhead were tagged with CWTs at Nimbus Hatchery in any year.

Fish entrained into the water export system were sampled at the SWP and CVP fish facilities at regular intervals by diverting the entire fish salvage flow into a separate holding tank. All steelhead in each sample were counted and measured. Although counts are typically used to estimate total salvage by multiplying the observed number of fish by the total minutes pumping divided by the sample length, counts were not expanded for this analysis.

Relationships between median annual transit times and Delta outflow were analyzed by simple correlation. Daily Delta outflow was obtained from DAYFLOW (CDWR 1986) and were averaged for the months of January and February of each year.

Results

Of the two hatcheries releasing CWT steelhead from 2000-2003, more salvaged steelhead came from Feather River Hatchery each year (Table 1). In 2003, almost 92% of CWT steelhead salvaged at the fish facilities were from the Feather River Hatchery (Table 2). In 2004, the first year that Mokelumne steelhead were tagged, about 91% of salvaged hatchery steelhead were from the Mokelumne hatchery and no CWT steelhead from the Feather River Hatchery were salvaged.

Median annual transit days from release to salvage ranged between 28.5 days and 75.0 days for Coleman Hatchery steelhead and between 38.5 days and 55.0 days for Feather River Hatchery steelhead (Table 3). Annual median transit time was greatest in 2003, especially for steelhead released from Coleman Hatchery. In 2004, median transit time was 25.0 days for steelhead released in the Mokelumne River. Transit times for individual fish varied substantially, from 6 to 409 days. Two fish salvaged in 2000 had transit times greater than 365 days, indicating possible overwintering.

Table 1 Annual numbers of hatchery-released CWT steelhead salvaged at CVP and SWP fish facilities.

Sample Year	Coleman	Feather	Mokelumne	Total
2000	44	55	N/A	99
2001	19	52	N/A	71
2002	14	20	N/A	34
2003	9	97	N/A	106
2004	9	N/A	93	102
Total	95	224	93	412

Median annual transit times for Coleman Hatchery steelhead were weakly positively correlated with January-February mean Delta outflow ($r = 0.12$) and median annual transit times for Feather River Hatchery steelhead also were weakly positively correlated with January-February mean Delta outflow ($r = 0.34$).

In 2004, Mokelumne River steelhead took about 5 days longer, on average, to arrive at the SWP facility than the CVP (Table 4), but for salvaged steelhead from other hatcheries there was no discernable trend in transit times between the two facilities.

Table 2 Annual percent of salvaged CWT steelhead from each hatchery.

Sample Year	Coleman	Feather	Mokelumne	Total
2000	44.4	55.6	N/A	100.0
2001	26.8	73.2	N/A	100.0
2002	41.2	58.8	N/A	100.0
2003	8.5	91.5	N/A	100.0
2004	8.8	N/A	91.2	100.0

Table 3 Median number of days between release and salvage at fish facilities.

Salvage Year	Coleman	Feather	Mokelumne
2000	42.0	46.0	N/A
2001	28.5	38.5	N/A
2002	46.0	43.0	N/A
2003	75.0	55.0	N/A
2004	31.0	N/A	25.0

Table 4 Median transit days for CVP and SWP facilities

	CVP			SWP		
Salvage Year	Coleman	Feather	Mokelumne	Coleman	Feather	Mokelumne
2000	34.0	49.5	N/A	43.0	46.0	N/A
2001	28.0	40.5	N/A	29.0	38.0	N/A
2002	38.0	38.5	N/A	47.0	49.0	N/A
2003	80.0	57.0	N/A	69.5	54.0	N/A
2004	31.0	N/A	23.0	26.5	N/A	28.0

Discussion

The salvage of two large fish (409 mm and 333 mm), over a year after release, indicates the possibility that there may be overwintering or residence in the Delta or its tributaries. A second possibility is that these fish were returning adult spawners. Other fish larger than 300 mm may have been salvaged after 365 days from release, but they would not have been sacrificed due to a change in protocol after 2000. Also, 20 of the 308 steelhead salvaged from 2000 to 2003 had transit times greater than three months indicating that some steelhead may be using the Delta for rearing habitat.

Differences in Delta outflow did not explain differences in annual median transit times for steelhead released from Coleman and Feather River hatcheries. A more detailed analysis of relationships between transit time and flow, water exports, and other water conveyance variables may be fruitful.

Steelhead did not take longer to arrive at the SWP facility than at the CVP facility, since there was no discernable trend in the median annual transit. Longer travel times to the SWP might be expected, since Clifton Court Forebay presents an additional obstacle for entrained juvenile steelhead.

The close proximity of the Mokelumne River Fish Installation to the fish facilities resulted in quicker transit of fish released from that hatchery, generally less than 4 weeks.

The relatively low and variable proportion of steelhead currently marked at Central Valley hatcheries results in a lack of reliable data to use in understanding the vulnerability of steelhead from various parts of the central valley to SWP/CVP entrainment. At present, there are some hatcheries that do not use CWTs and others that use a proportion that varies annually, making it impossible to estimate the relative contribution of each hatchery to the fish salvage. A constant fractional marking program for Central Valley steelhead would permit an analysis of which hatchery steelhead were most vulnerable to entrainment at the salvage facilities. Under such a program, all or a constant fraction of steelhead released from Central Valley hatcheries would be uniquely marked with origin site, release date, and site information.

Potentially, hatchery releases and water management practices, particularly SWP/CVP exports, could be coordinated so that direct seaward emigration of smolts is maximized. CWT steelhead recoveries in 2004 show the potential of water export facilities to influence the migration of hatchery fish, particularly for those released from Mokelumne River Hatchery. Water export reductions dur-

ing and after Mokelumne River hatchery releases could benefit smolts by reducing the probability of their appearance at the Delta export facilities.

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Subregions of the Sacramento-San Joaquin Delta: Identification and Use

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Introduction

In recognition of the pervasive importance of phytoplankton in ecosystem processes, variables related to phytoplankton abundance and growth rate have been measured in San Francisco Estuary monitoring programs extending back to the 1960s, including biomass as chlorophyll *a*, taxonomic composition, vertical light attenuation and a variety of nutrients. Currently, these variables are measured in the upper estuary (Sacramento-San Joaquin Delta through upper San Pablo Bay) under the auspices of the Interagency Ecological Program (IEP) for the San Francisco Estuary. The IEP has among its goals the determination of trends in ecological resources and the factors underlying these trends. The IEP Environmental Monitoring Program includes both routine monitoring and special studies to achieve these goals.

It is important to estimate phytoplankton biomass and production trends at the system-wide scale, in addition to specific locations: System-wide averages provide indices of ecosystem services such as net carbon sequestration and food production for fisheries. They also enable the construction of mass balances that improve understanding of processes at the ecosystem scale. Ideally, the average should be determined from a probability-based sample, i.e., a random sample in which every location has a known probability of being sampled. Only a probability-based sample eliminates the risk of bias from subjective sample selection and is independent of an assumed model. Moreover, valid confidence levels can be placed on the estimates, enabling the objective detection of system-wide trends. The historical collection of stations in the IEP Environmental Monitoring Program represent a judgment sample, however, not a probability sample. The stations were chosen to cover the spectrum of water quality behavior in the estuary, but not necessarily to provide system-wide statistics with confidence levels.

System-wide averages must therefore be determined by assuming some kind of model. For example, a station or subgroup of stations is assumed to be representative of a preselected homogeneous subregion within the estuary. The averages for the subregions are then weighted by the respective subregion areas or volumes, depending on the variable, and combined to provide a system-wide estimate. Such a stratification of the estuary can provide a more precise estimate of the system-wide mean if the within-subregion variability is reduced relative to the between-subregion variability (Thompson 1992). If stations do fall into relatively homogeneous groups that are stable in time, then it might also be possible to identify redundancy among the stations and reduce their overall number. Jassby and Cloern (2000) made system-wide estimates of primary productivity, chlorophyll *a*, and $_{TSS}$ in this manner, based on subregions suggested by Lehman (1996). Although confidence intervals and bias cannot be evaluated without a probability sample for comparison, any division of the stations into purported homogeneous subregions should exhibit consistency with the data. How well do the historical data in fact support this concept of relatively homogeneous subregions that are stable over time?

A second reason to divide the estuary into subregions is for the purpose of understanding local variability processes. In this case, values for stations within the same region are not necessarily similar in magnitude. Rather, they are correlated in time, a sign that they are responding to the same underlying variability mechanisms. Stations within such subregions can be viewed as equivalent or redundant for some data analysis and modeling activities. This implies that, instead of needing to understand vari-

ability processes at each of a large number of stations, we may be able to choose the best single record from each subregion and focus attention on a much smaller set of stations. Can we indeed identify subregions, each of which represents a portion of the estuary that is responding over time in a uniform way to stressors?
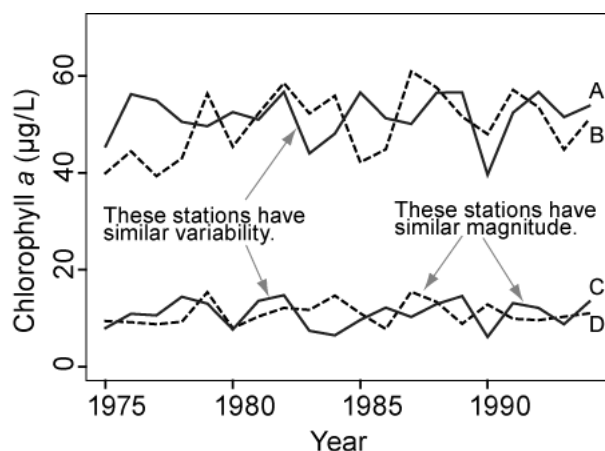

Figure 1 Chlorophyll *a* time series at four hypothetical estuarine stations. A is correlated with C, and B is correlated with D, in time.

Four hypothetical time series illustrate the distinction between these two ways of choosing subregions (Figure 1). Series A and B have similar magnitudes of about 50 µg/L chlorophyll *a*, while series C and D have magnitudes of about 10 µg/L. Based on magnitude, A and B should fall into one subregion, whereas C and D should fall into a second subregion. It might be appropriate to average stations within each of these subregions for estimating system-wide averages. On the other hand, A and C are correlated in time, as are B and D. Based on temporal variability, A and C should fall into the same subregion, whereas B and D should fall into a second separate subregion. Stations within these latter subregions appear to be responding to the same variability mechanisms. As elaborated below, this information can be used to help identify the processes underlying variability, as well as to simplify subsequent analyses.

Here, we explore these different ways to identify and make use of subregions in the Sacramento-San Joaquin Delta using long-term phytoplankton and water quality data collected by the IEP Environmental Monitoring Program. Similar analyses could be done for other IEP data sets. We gratefully acknowledge support for this research from the California Bay-Delta Authority (ERP-02-P33).

Methods

Data Preparation

Analyses are confined to data collected consistently at long-term monitoring stations maintained by the IEP Environmental Monitoring Program in the Sacramento-San Joaquin Delta and Suisun Bay (Figure 2). The analysis of spatial structure requires a large number of stations for many years. Because stations were dropped and added over the years, only a subset of the monitoring data can be used for this purpose. First, replicate measurements were averaged, and the time series for each station was constructed from monthly median values. We then explored the patterns of missing chlorophyll a data for all stations for the period 1969-2002 and chose a total of 25 stations with sufficient data for analysis (Figure 3). The percentage of missing data for these stations ranged from 7% for C7 to 45% for MD7. In order to avoid large gaps in the time series, the analysis of data collected at these 25 stations had to be confined to the period 1975-1995. This reduced the missing data to between 1% (C7) and 13% (MD7).
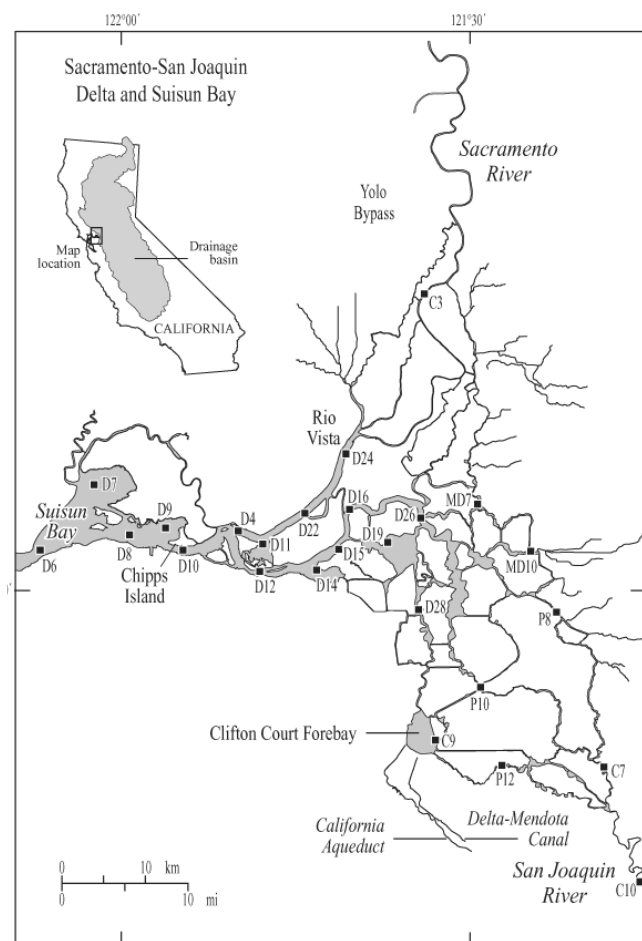
Figure 2 The Sacramento-San Joaquin Delta and Suisun Bay, showing the locations of monitoring stations used in this study (modified from a map by Jeanne DiLeo, US Geological Survey).
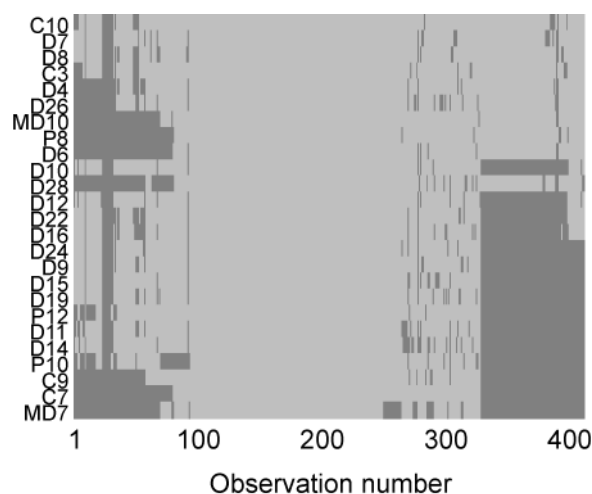
Figure 3. Monitoring stations sorted by the proportion of missing monthly chlorophyll a data for the period September 1969-December 2002, a total of 412 months. Dark gray, data missing; Light gray, data available.

Subregions for Spatial Averaging

A traditional approach to the problem of identifying homogeneous subregions is cluster analysis. Cluster analysis has been widely used in ecology and also widely criticized. A major problem is the failure to determine the significance especially of the finer clusters. This problem can be addressed by model-based clustering, which may use traditional hierarchical agglomeration methods but also assumes that the data are generated from a mixture of underlying probability distributions (Banfield and Raftery 1992). This method was used here to investigate how stations cluster on the basis of chlorophyll *a* and other water quality variables.

The basic clustering method starts with each station forming its own group. At each subsequent step, two groups are chosen for merging so that the sum of within-group sums of squares increases minimally (Ward 1963). Although the algorithm can use criteria other than one based on sum of squares, this classic criterion is most suitable for decreasing the uncertainty in system-wide averages based on cluster (strata) means. A model-based form of the clustering procedure was used, which assumes that data are generated from a mixture of underlying probability distributions (Banfield and Raftery 1992). This enables calculation of the approximate weight of evidence (AWE) for k clusters. The AWE, in turn, is an estimate of $2 \log B_k$, where B_k is the odds for the model defined by k clusters against the model defined by a single cluster. The

value of k that maximizes AWE is therefore the one for which there is the most evidence.

As a check on the model-based clustering, we also used a completely different clustering method. In a graded environment like an estuary, stations are more apt to make slow transitions from one cluster to another, and conventional methods that assign each station to exactly one cluster—known as *crisp clustering*—may be inherently inappropriate. *Fuzzy clustering* methods, on the other hand, allow fractional membership in many clusters, allowing a more realistic representation of relationships among stations (Kaufman and Rousseeuw 1990).

For each object (station) i and cluster v, a membership u_{iv} indicates how strongly object i belongs to cluster v. For k clusters, memberships must satisfy the following conditions:

$$
\begin{aligned}
u_{iv} \geq 0 \qquad & i = 1,\ldots,n \text{ and } v = 1,\ldots,k \\
\sum_{v=1}^{k} u_{iv} = 1 \quad & i = 1,\ldots,n
\end{aligned}
\tag{1}
$$

The memberships are then defined through minimization of an objective function. Each station i is assigned to the cluster v in which it has the highest membership u_{iv}. This produces a crisp clustering that can be compared directly with the model-based clustering. Note that a given cluster v may not have the highest membership u_{iv} for any object i, in which case that cluster will not contain any crisply clustered stations. A quantity known as the *silhouette width* describes how well each station is associated with its assigned cluster, ranging from -1 (badly classified) to +1 (well classified). The number of clusters k was determined by maximizing the overall average silhouette width.

Subregions for Studying Processes

We also identified subregions within which the same processes appear to be operating because of a high correlation among the stations within the subregion. We took an exploratory approach used in meteorology and oceanography, in which the variability for each station is viewed as the outcome of a number of underlying processes or *modes* of variability (Preisendorfer 1988). The method is based on principal component analysis, viewed as a mathematical technique with no underlying statistical model

(Chatfield and Collins 1980). Jassby (1999) describes the approach in the context of several ecological applications.

Let $\mathbf{X} = [\mathbf{x}_1,\ldots,\mathbf{x}_p]^{\mathrm{T}}$ be the $p \times n$ observation matrix for a p-dimensional time series of length n with mean $\overline{\mathbf{x}}$. In this study, each $\mathbf{x}_i$ represents the series of monthly observations at station i. Let $\mathbf{S}$ be the sample covariance matrix, and $\mathbf{A}$ be the $p \times p$ matrix of eigenvectors of $\mathbf{S}$. Then the p principal components of $\mathbf{X}$ are given by

$$
\mathbf{Y} = \mathbf{A}^{\mathrm{T}}(\mathbf{X} - \overline{\mathbf{x}})
\tag{2}
$$

These linear combinations of the original observations have the property that the first principal component accounts for as much of the variation (given by its corresponding eigenvalue) in the original data as possible, the second accounts for as much of the remaining variation as possible, and so on. $\mathbf{A}^{\mathbf{T}}$ provides the *coefficients* that define the influence of each station on each component. For the rth observation (column) $\mathbf{w}_r = [x_{1r},\ldots,x_{pr}]^{\mathrm{T}}$ of $\mathbf{X}$, Equation 2 becomes

$$
\mathbf{y}_r = \mathbf{A}^{\mathrm{T}}(\mathbf{w}_r - \overline{\mathbf{x}})
\tag{3}
$$

where $\mathbf{y}_r$ is the component score or *amplitude* for that observation. In this study, the correlation rather than covariance matrix is used, to prevent the results from being dominated by a few stations with very high measurements. Equation 3 is then valid only after standardizing $\mathbf{w}_r - \overline{\mathbf{x}}$ so that each variable has unit variance.

Because of the way principal components are chosen, most of the information contained in $\mathbf{X}$ may be contained in just the first few m components. Here, the number of important principal components is determined with a Monte Carlo technique known as *Rule N* (Overland and Preisendorfer 1982). Rule N involves computing the eigenvalues of a large number of uncorrelated $n \times p$ data sets. Each experimental eigenvalue is then compared with the 0.95 quantile of the corresponding simulated eigenvalues.

The inverse transformation of Equation 2 can then be written approximately as

$$
\mathbf{X} = \mathbf{A}_m \mathbf{Y}_m + \overline{\mathbf{x}}
\tag{4}
$$

where $\mathbf{A}_m$ is the truncated $p \times m$ matrix of eigenvectors and $\mathbf{Y}_m$ is the $m \times n$ matrix of the first m principal components. The reduced set of principal components is rotated to find a new set of components (no longer principal) with simple structure, in which each variable has high coefficients for as few components as possible (Richman 1986). Here, the varimax method is used to attain simple structure.

Rule N in this spatial context reveals the number of underlying components or modes of variability that together combine to produce most of the overall spatial variability. The mode coefficients define the influence of each station on the mode. Each mode also has a set of amplitudes associated with it that describes the strength of the mode for each month. The number of modes, the coefficients, and the amplitude time series together provide strong constraints on the underlying mechanisms while also providing clues for their identity.

Problems arise when we need to identify subregions for more than a single variable. Current estimates of phytoplankton production from water quality data require multiple variables, at least chlorophyll *a* and total suspended solids (TSS) in addition to, of course, surface irradiance (Jassby et al. 2002). Possibly inorganic nitrogen and temperature will need to be incorporated in the future. Even if subregions exist for these variables individually, the subregions will not necessarily coincide. To investigate this issue, we applied the regionalization procedure separately to both chlorophyll *a* and TSS.

Results

Subregions for Spatial Averaging

The AWE for model-based clustering of the monthly chlorophyll data attained a maximum for four clusters (Figure 4). Three of these clusters, however, consisted of a single station, all in the south Delta (Figure 5). Thus, model-based cluster analysis of chlorophyll adds little to our understanding of station groupings based on month-scale variability, as these three stations are easily identified simply on the basis of their much higher chlorophyll values. Similar results were obtained for every water quality variable tested. We found that AWE was maximal for k = 1 in the case of water temperature, turbidity, TSS, ammonium, and total phosphorus. The only exception was electrical conductivity. The AWE reached a maxi-

mum at k = 4 for conductivity, dividing the stations into four clusters along the estuarine axis: (1) D6, D7 and D8; (2) D9 and D10; (3) D4, D11, and D12; and (4) all other stations. Again, this adds little to our understanding of station groupings, especially as almost all Delta stations fall into a single cluster.
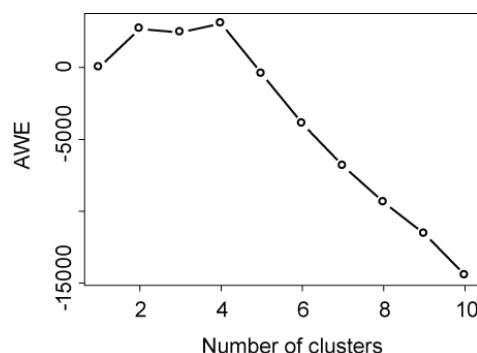
Figure 4 Approximate weight of evidence (AWE) plot for model-based hierarchical clustering of the chlorophyll *a* data. AWE is maximized for four clusters, for which there is therefore the most evidence.
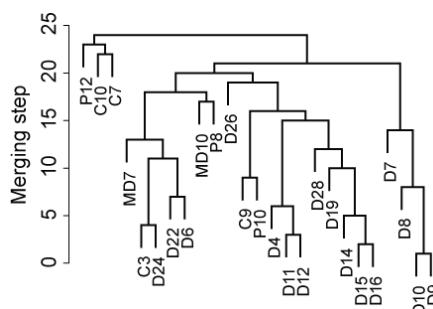
Figure 5 Clustering tree for model-based hierarchical clustering of the chlorophyll a data. The south Delta stations (P12, C10, C7) clearly stand apart from the remaining stations.

The average silhouette width for fuzzy clustering of the monthly chlorophyll data attained a maximum at 0.78 for only two clusters. The smaller of these consisted of stations C7, C10, and P12, the same three south Delta stations distinguished by the model-based clustering. Increasing the number of fuzzy clusters to three did not change the crisp clustering or the average silhouette width noticeably, because none of the stations had their highest membership in the third cluster. Increasing the number of fuzzy clusters still further resulted in a distinctly lower average silhouette width, as well as an increasing number of badly-classified stations (silhouette width close to zero

or negative). The fuzzy clustering results for other water quality variables were also consistent with the model-based clustering, although the first two and last two clusters for electrical conductivity were combined and D4 was poorly classified. The results therefore point to a clear differentiation only between Suisun Bay and the Delta.

Subregions for Studying Processes

Chlorophyll

Application of Rule N to the chlorophyll a time series suggested that there were three important principal components. These first three components accounted for 68% of the total variability in the chlorophyll a data. The modes themselves are conveniently described with the use of starplots (Figure 6). Each station is symbolized by three axes proportional to the station coefficients for each mode. Each station symbol is colored according to the mode for which it has the highest coefficient. The modes are designated as Delta, river, or Suisun modes, depending on where their corresponding stations are primarily located. These modes accounted for 32%, 14%, and 22%, respectively, of the total variance. D6 and MD10 have coefficients less than 0.2 for all modes; although they may be associated primarily with a single mode, the association is a relatively weak one.

The 3 amplitude time series for the modes represent a simplification of the 26 time series for individual stations (Figure 7). The Delta mode amplitudes exhibit no obvious pattern or long-term change during 1976-1995. The river mode amplitudes are high during the 1976-1977 period and then relatively high again toward the end of the 1987-1992 drought. The Suisun mode amplitudes show a distinct drop after 1986.
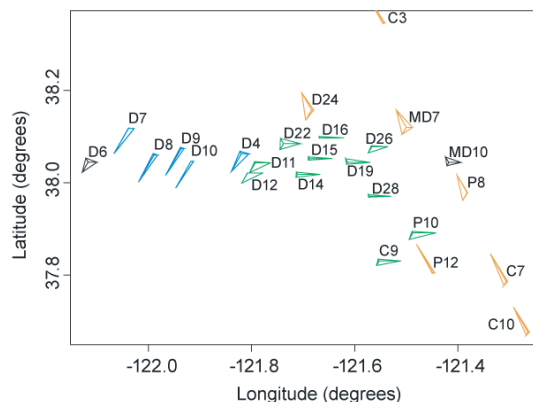
Figure 6 Star plot of the chlorophyll a mode coefficients for each station. Each star has three axes, and each axis represents the size of the coefficient for the corresponding mode. Each station is associated by color with the mode for which it has the highest coefficient (stations in black have no strong associations: all coefficients are less than 0.2). A narrow shape indicates association mostly with only one of the modes, whereas a broad shape indicates association with two or three modes. A large star in one or more directions indicates that the station's behavior is strongly associated with at least one of the modes; a small star indicates that the station is not strongly associated with any mode and its variability is largely due to other processes. Green, Delta mode; orange, river mode; blue, Suisun mode.
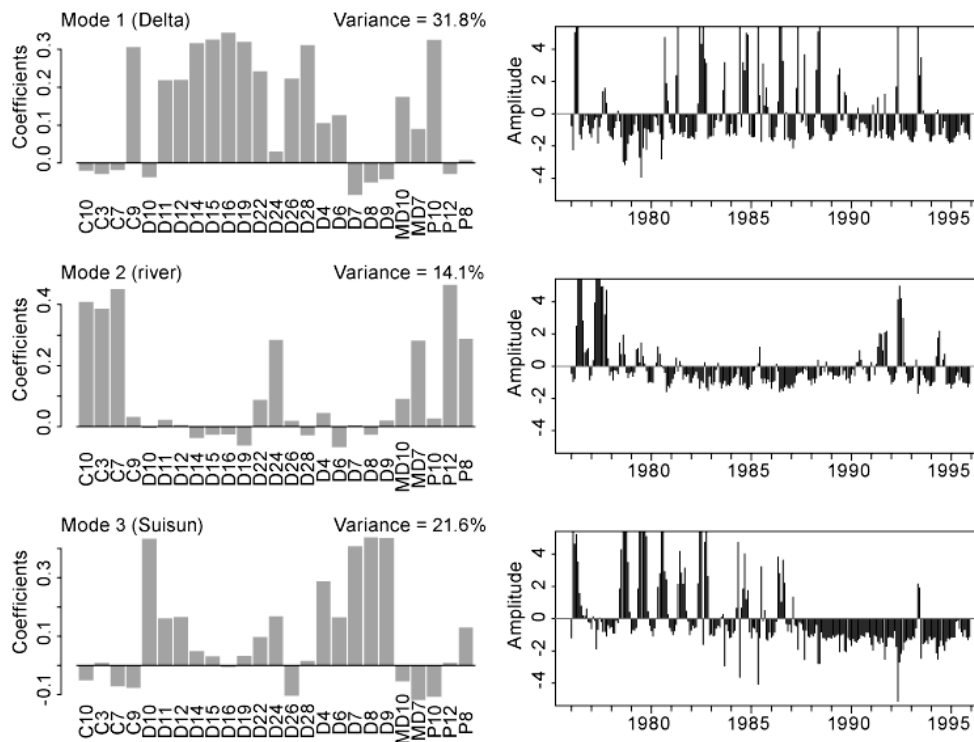
Figure 7 Coefficients and amplitude time series for each chlorophyll *a* mode. Coefficients describe the importance of the corresponding mode for the variability of each station. The amplitude time series describes the strength of the corresponding mode over time. The variance associated with each of mode is also shown.

Total suspended solids

Vertical light attenuation is another variable of great importance for primary productivity. TSS is, in turn, the best guide in the dataset to long-term variability in attenuation. Direct measurements of attenuation coefficient number only half of those for TSS. There are more measurements for Secchi depth than TSS, but Secchi depth has very low precision in the Delta, where the median value is less than 0.5 m. There are even more measurements for turbidity, but like Secchi depth, turbidity is a relatively low-precision measurement.

Rule N applied to the TSS data suggested three significant components once again. Together, they accounted for 65% of the total variability. The mode with the most stations extends from Suisun Bay through the western Delta (Figure 8). To the south and east lies another mode with six stations. The third and final mode consists only of C7 and C10 in the San Joaquin River, although P12 also has a high association with this mode. Seven stations have weak associations (coefficients less than 0.2) with all modes, including C3, D7, D9, D14, D19, D22, and MD7.

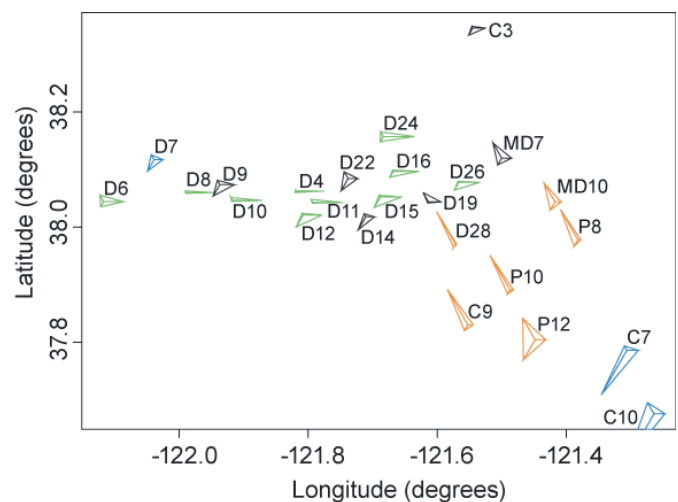The modes are designated north, intermediate, and south modes for convenience.

Figure 8 Star plot of the TSS mode coefficients for each station. See Figure 6 for details. Green, north mode; orange, intermediate mode; blue, south mode.
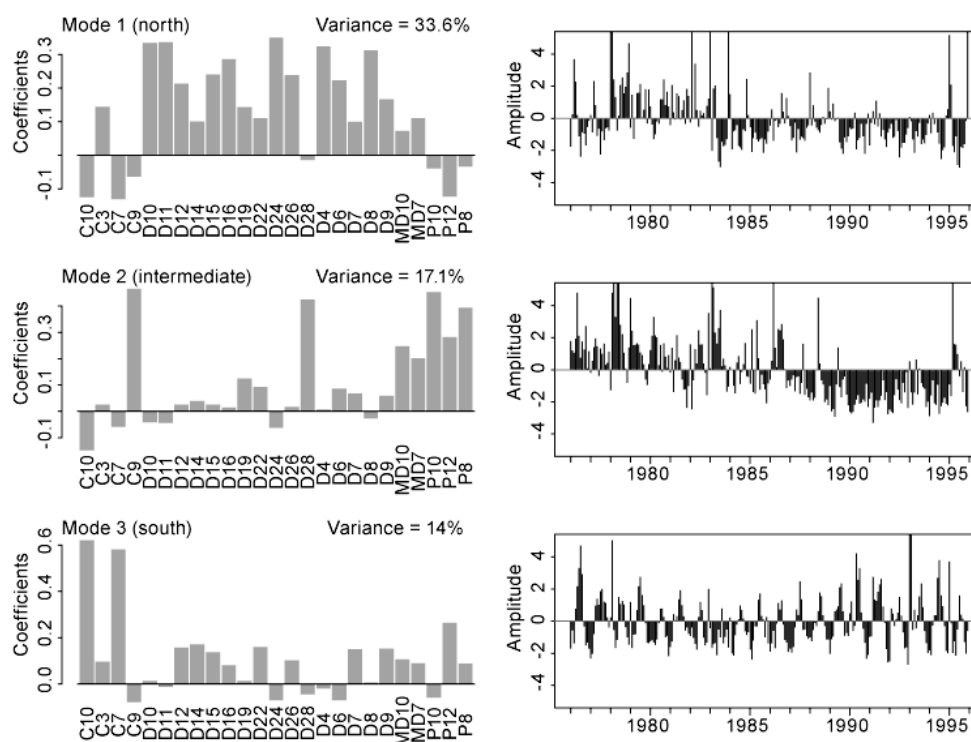
Figure 9. Coefficients and amplitude time series for each TSS mode. See Figure 7 for details.

The north mode accounts for more than a third of the total variability (Figure 9). A prominent feature of the corresponding amplitude time series for this mode is a clear drop toward the end of the 1982-1983 El Niño-Southern Oscillation (ENSO) event, from which there is no recovery at least through 1995. The intermediate mode, on the other hand, exhibits more of a response to water year type. The amplitude is high for 1983 and declines during the extended drought from 1987-1992, although behavior during the dry 1976-1977 ENSO is a prominent exception. The remaining south mode shows no convincing long-term pattern, but rather is dominated by seasonal variability.

Discussion

Subregions for Spatial Averaging

Stable consistent station clusters do not appear to exist for the water quality variables examined here, aside from the somewhat trivial cases found for chlorophyll a and conductivity. The landward estuary, including Suisun Bay and the Delta, is certainly a heterogeneous environment that can be considered patchy and composed of more

or less homogeneous subregions at any given instant. But it is also a dynamic environment, highly responsive to river flows, tides, and winds. A particular pattern is apparently too short-lived to survive the seasonal and annual variability represented by these monthly time series. The model-based cluster tree does represent real affinities among stations, and ranks them in importance as well (Figure 5). But the lack of statistical significance for all, or all but a few, branches means that the station groupings cannot be counted on for any individual sampling event, or any individual system-wide estimates. Perhaps this is to be expected. These monitoring stations represent a judgment sample originally chosen to capture the gradients in the upper estuary efficiently. The lack of stable consistent station clusters implies that station configuration is indeed efficient in the sense that duplication has been avoided.

How, then, can system-wide averages be estimated? The requirement for historical continuity and the cap on program costs (Hymanson and Müller-Solger 2003) means that a probability-based station network cannot be considered at present without sacrificing temporal resolution. Moreover, random sampling would be difficult to implement, as many areas of the Delta are relatively inac-

cessible to well-equipped boats or land vehicles. Schemes for interpolating between stations may yield more realistic system-wide estimates than the assumption of distinct subregions, but developing a stable, consistent interpolation scheme suffers from the same problem as developing a stable, consistent subregion scheme. As our understanding of water circulation patterns improves, it is hoped that an interpolation scheme can be developed based on river flows, exports, and other flow-related factors. This has motivated an increased emphasis on continuous sampling stations at or near flow stations. For the most important phytoplankton-related variables, namely, chlorophyll *a* and TSS, satellite imagery also holds promise as a way to make system-wide estimates. But in the interim, there is little opportunity to improve on existing approaches that divide up the estuary into subregions based on "physical and ecological conceptual models of the estuary," including geometry, bathymetry, transport processes, hydrologic inputs, and habitat types (Müller-Solger and Hymanson 2003). Fortunately, these proposed subregions resemble the clusters based on chlorophyll *a* data, even though the latter was only one of many pieces of information involved in the choice of subregions.

System-wide estimates, although essential for certain purposes such as mass balances, may not be the most important objective for allocating sampling resources. As far as trends are concerned, it is still possible and perhaps more informative to identify these at individual stations and use their collective values rather than a scalar system-wide index. In such a spatially-complex system, behavior at the system-wide level may be of little use in understanding the important variability processes, many of which seem to take place at smaller spatial scales, as discussed in what follows.

Subregions for Studying Processes

The application of principal component analysis to the water quality data yields subregions within which the stations are varying through time in a similar manner. Cluster analysis also does this indirectly, but it does so by finding stations that have similar magnitudes at any time. The principal component analysis is thus less restrictive in a certain sense and more likely to arrive at meaningful subregions. This same lack of restrictiveness means, however, that the resulting subregions cannot be used for improving system-wide estimates. For example, C3 on the Sacramento River and C10 on the San Joaquin River both belong to the river mode, but their chlorophyll *a* values

usually differ by at least an order of magnitude. There is nothing to prevent us from further subdividing the subregions chosen by principal component analysis on the basis of magnitude or other criteria in order to refine our understanding of mechanisms. For example, a division of the river subregion into a Sacramento and San Joaquin portion is warranted because of the huge difference in nutrient concentrations, which may have second-order effects on variability that are not detected by the analysis.

The starplots of station loadings remind us that stations should not be identified with single mechanisms (Figure 6). D24, for example, has relatively high coefficients for both the river and Suisun modes. They also demonstrate that there can be gradual rather than abrupt change in the importance of variability processes from one subregion to the next. This is especially obvious at the boundaries of the Delta and Suisun modes. One of the strengths of this type of analysis is that it preserves and reveals the dependence of individual station behavior on multiple processes. Like fuzzy clustering techniques, it does not posit the artificially abrupt changes between subregions that can result from other classification procedures.

The presence of three significant chlorophyll *a* modes implies that our understanding of most of the variability can be reduced to understanding the variability at characteristic stations, namely, those with high component coefficients for one mode and low ones for the remaining modes. This offers an opportunity to reduce the number of monitored stations for investigating the determinants of change. Transitional stations like D24 have time series that may appear unique, but this analysis shows that they can be understood as a combination of mechanisms that have a more "pure" expression at other stations (e.g., C3 and D10). Although there is no objective criterion for deciding the point at which a station becomes redundant, the coefficients help to choose a given number of stations from among all available stations to maximize information about variability. Characteristic stations should be chosen based not only on having high component coefficients for only one mode, but also on historical data availability. This allows use of longer-term data than possible when analyzing a large number of stations simultaneously. In the case of chlorophyll *a*, example characteristic stations are D28A for the Delta mode, C10 for the river mode, and D7 or D8 for the Suisun mode. These offer the best combination of fidelity to the mode (Figure 7) and data availability (Figure 3).

For some stations, however, the association with any mode is so low that an understanding of the modes is insufficient to understand behavior at the station. For example, D6 and MD10 have low coefficients for all modes. These stations may need to be addressed individually to understand the nature of their variability.

The TSS modes overlap but do not coincide with the chlorophyll *a* modes. They make it clear that no division into subregions will be suitable for all water quality variables simultaneously. Each variable of interest must be subjected to this analysis independently. Conveniently, though, D28A is an appropriate representative for the intermediate mode, C10 for the south mode, and D8 for the north mode (Figure 9). The same network of key sites can thus be used for both chlorophyll a and TSS.

Although it is beyond the scope of this article to examine the mechanisms underlying the different modes, it is worthwhile to note how informative the amplitude time series are regarding these mechanisms. For example, the amplitude for the Suisun chlorophyll *a* mode drops after 1986 and does not recover through 1995, suggesting that this mode is driven by the invasion of the clam *Potamocorbula amurensis* (Figure 7; Alpine and Cloern 1992). On the other hand, the amplitude for the north TSS mode drops during the 1982-1983 ENSO event and remains low, suggesting a long-term impact from the unusually high Sacramento River flows. A more detailed examination of the key stations is now underway to understand better the mechanisms behind each mode.

While continued monitoring at these key sites offers an ongoing opportunity to learn more about large-scale variability processes for certain water quality variables, the larger number of stations spread out across the Delta remains critically important. In order for any station network to represent the response of the broader ecosystem to stressors, three demanding conditions must be fulfilled (Jassby 1998): (1) some sites in the network must encounter the stressor; (2) some of these latter sites must have the critical features that cause a response to the stressor; and (3) the background variability must not disguise the response. The subregions and set of key stations found here reflect the effect of climatic and biological stressors that occurred in the past. There is no guarantee that the same key stations can adequately fulfill these three conditions for detecting new stressors. Moreover, there is often a need for observing and understanding variability related to environmental conditions at smaller spatial scales or at

specific locations other than these stations. Thus, ongoing comprehensive station coverage of the Delta is essential.

Conclusions

- A probability-based water quality sampling network in the Delta is not feasible because of cost, logistical problems, and the need for historical continuity. Because the existing station network is a judgment sample, not a probability-based sample, system-wide averages must be determined by assuming some kind of model, e.g., a division of the estuary into homogeneous subregions. Confidence intervals for system-wide averages, however, cannot be determined.

- Although relatively homogeneous subregions of the estuary with respect to water quality may exist on any given sampling day, cluster analysis of water quality measurements suggests that these subregions are not stable in time. System-wide averages that are based on averaging measurements in fixed subregions can therefore be biased.

- Interpolation among existing stations may become possible as hydrodynamic models and/or satellite imagery improves. Without interpolation, we have to assume a division into subregions, no matter how tenuous, based on current physical and ecological conceptual models of the estuary.

- Subregions based on stations exhibiting similar temporal variability in water quality, rather than similar magnitudes, can be identified.

- Although such subregions may be different for different variables, they can overlap, in which case it may be possible to pick a few stations that simultaneously represent multiple variables. For example, D8, D28A, and C10 perform this function well for both chlorophyll *a* and total suspended solids, although other combinations are also possible.

- These subregions offer a way to choose a small number of stations for analyzing long-term variability processes. Special effort should be

made to preserve continuity of measurement at these key stations.

- Although the identification of key stations simplifies the quest to understand historic estuarine water quality patterns, comprehensive station coverage of the Delta remains necessary to detect new processes and to examine variability at smaller spatial scales or specific locations.

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Investigating the Mechanisms Underlying the Relationships Between Abundance of Estuarine Species and Freshwater Flow

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Introduction

Freshwater flow and tides are generally the key variables influencing patterns of circulation in estuaries. These variables are therefore also key influences on chemical and biological variability, although at different time scales. While tides provide most of the kinetic energy for mixing and advection in estuaries, freshwater flow sets up the salinity gradient thereby interacting with tides to

determine how that mixing and advection play out. In addition, freshwater flow determines the rate of delivery of materials and organisms to the estuary. Freshwater flow varies at all time scales, including daily, seasonal, and inter-annual scales, important biological time scales that can result in interactions between freshwater flow and organismal abundance or position (Kimmerer 2004).

The influence of freshwater flow has received increased attention recently in the scientific literature (Montagna et al. 2002, Kimmerer 2002a, b). This interest is driven to some extent by concern over changes in freshwater flow due to climate change (Knowles and Cayan 2003). The early literature on effects of flow on the biota of estuaries contained several references about the San Francisco Estuary (Turner and Chadwick 1972, Peterson et al. 1975, Stevens 1977, Arthur and Ball 1979, Cloern et al. 1983). Positive effects of freshwater flow on the abundances of various fish species were described by Stevens and Miller (1983), Armor and Herrgesell (1985), and Hatfield (1985).

Jassby et al. (1995) analyzed monitoring data collected by the Interagency Ecological Program to determine the response of various fish and shrimp species to freshwater flow. They used the distance up the axis of the estuary to the tidally-averaged 2 psu isohaline ("X2") as a measure of the physical response of the estuary to flow. Later analyses updated these relationships with more recent data and determined that, although the intercepts of some had changed, slopes of the relationships had not changed in 2 or more decades (Kimmerer 2002a). Thus, these relationships appear robust to environmental changes due to other factors such as the introduction of the clam *Corbula amurensis* (see note by Jan Thompson, this issue, regarding the name change).

X2 is currently being used in a standard to protect estuarine habitat. Specifically, during February - June flow is controlled to place X2 at or seaward of one of 3 control points, depending on available water. Because of the high economic value of water needed to meet the standard at some times (Kimmerer 2002b), there is considerable interest in making the standards more efficient or effective. To determine whether this is possible will require an understanding of the mechanisms underlying the relationships between abundance or survival of fish and flow or X2.

We are working with several other researchers using CALFED Ecosystem Restoration Program funding, and with the Estuarine Ecology Team, to conduct some initial model investigations and develop a research plan to determine which mechanisms are important for which species. This paper briefly reviews the status of mechanisms underlying the "fish-X2" relationships, providing a summary of some of the analyses done to date. Potential mechanisms underlying these relationships were first outlined in a brief technical report (EET 1997), and some of the more likely ones were discussed by Kimmerer (2002b, 2004). Although numerous mechanisms are possible, we focus here on those that seem most consistent with the currently available data, and do not attempt to explore all possibilities . We then present an approach to resolving these mechanisms through a program of research.

The physical basis of X2

The physical effects of changing freshwater flow have been explored in several recent papers (Monismith et al. 2002, Kimmerer 2002b, 2004). There is no need to reiterate those results, except to highlight a few key points. First, changing freshwater flow changes many conditions in the estuary, complicating the search for mechanisms (Kimmerer 2004 Figure 37). Second, X2 varies with freshwater flow either as a logarithmic function (Jassby et al. 1995) or a power function (Monismith et al. 2002), the latter having a clear physical interpretation. Third, the response of the estuary seaward of the Low-Salinity Zone (LSZ) depends on the interaction between compression of the longitudinal density gradient and water depth to produce stratification and landward salt flux. These conditions are fundamentally different from those occurring in freshwater; thus, the LSZ represents a boundary between two fundamentally different physical regimes (although not necessarily a boundary for biota).

Different researchers have different concepts of X2. Often it is seen either as simply a surrogate for freshwater flow, or as a measure of habitat for certain fish species. The first definition implies that it is a superfluous measure. The second implies that we know more than we actually do about what constitutes habitat and how its availability varies with X2.

An alternative view is that X2 is a measure of the physical response of the estuary to freshwater flow. In this context it provides us with several useful pieces of information. First, X2 lags behind changes in freshwater

flow and therefore provides a measure of the time scale of responses of the estuary to changing flow. Second, it provides a geographic context by which changes in flow can be linked to changes in habitat conditions throughout the estuary. Third, it can be used to indicate where and when physical conditions in the estuary may undergo transitions from one state to another, particularly between stratified and unstratified. This is important to our understanding of the ecology of the estuary, because except in extreme cases of high flow, persistent stratification is uncommon (Monismith et al. 1996). Changes in freshwater flow affect the brackish to saline parts of the estuary largely by compressing the salinity gradient; thus, X2 is a more appropriate measure of physical conditions in the estuary than flow itself, even though flow is obviously the controlling variable.

Biological responses

The statistical characteristics of responses of estuarine biota to X2 have also been thoroughly explored (Kimmerer 2002a, b, 2004), and we briefly summarize them here. The responses to X2 are complicated by changes in the lower trophic levels of the estuarine food-web following introduction of the clam *Corbula amurensis* and its subsequent spread throughout the northern estuary in 1987. Most of the X2 relationships fall into two categories: those with a strong flow response, and those

with a strong downward step after 1987 (Table 1). Generally the nekton (fish and bay shrimp) were more responsive to flow whereas plankton responded to the step change.

The covariation of so many flow-related features of the estuary rules out a statistical analysis to compare alternative mechanisms. Nevertheless, the details of these relationships give some hints about the underlying mechanisms, Kimmerer (2002b) argued that the qualitative differences in relationships between plankton and nekton suggested that the X2 relationships of the nekton did not arise through a response of lower trophic levels to flow. A mechanism based on the response of lower trophic levels is not completely ruled out by this argument, particularly since the details of feeding interactions in the LSZ are poorly known.

Another important feature of the relationships is that they represent a dependence of the log of abundance or survival on X2; that is, abundance or survival increases exponentially with a seaward movement of X2. This will likely impose limits on the mechanisms that must be considered, in that some mechanisms may require unrealistic parameter values to achieve the rate of increase with flow that has been observed for some species.

Table 1. Summary of X2/flow effects on estuarine biota and effects due to a step change in 1987, presumably caused by the introduced clam *Corbula amurensis*. Except as noted data are through 2001. Statistically signficant effects only are noted; full details in Kimmerer (2002a, Table 4). Negative effect of X2 indicates positive effect of flow.

Response variable	X2 effect	Step change	Remarks
Chlorophyll	–	Down	
Eurytemora affinis	--	Down	Post-clam X2 effect spring only
Synchaeta bicornis	–	Down	
Neomysis mercedis	Mixed	Down	Negative before clam, positive after
Crangon franciscorum	Negative	--	
Starry flounder	Negative	Down	
Pacific herring	Negative	--	Survival index
American shad	Negative	Up	
Delta smelt	Mixed	--	Summer townet index; slope changed 1980-81
Longfin smelt	Negative	Down	
Sacramento splittail	Negative	--	
Striped bass	Negative	–	Survival index only through 1994

Our approach to examining the mechanisms is to use simple models where possible to extend our existing

knowledge and make predictions about how the mechanisms might work. If we can use models in such ways we

might be able either to rule out mechanisms, or determine a particular point in the estuary or in the life cycle where a key measurement might be used to assess the likelihood that the mechanism is operating.

General mechanisms

Our focus is on estuarine-dependent species of higher trophic levels, namely fish and the bay shrimp *Crangon franciscorum*. The mechanisms most likely differ among species because of their varied life histories and seasonal patterns of recruitment. Furthermore, some species may be affected by X2 or flow through more than one mechanism (Bennett and Moyle 1996). However, mechanisms can be grouped according to the life stage they operate on, and the requirements for each mechanism can be ruled out for some species (Figure 1).

Mechanisms that could apply to all species are related to four key aspects of the life cycle: transport, water clarity (relating to predator avoidance), food supply, and physical habitat. We discuss these in turn below, presenting analyses that help to focus on key aspects of each mechanism.

Mechanism T: Transport refers to the movement of fish from spawning to nursery grounds, either from upstream by river flow to the estuary and then by tidal flow, or from downstream by tidal flow and gravitational circulation. This mechanism could work in one of two ways: 1) Travel time decreases with seaward X2, and daily mortality during travel is higher than that at the destination; or 2) daily mortality during transport decreases with seaward X2. Here we estimate the form of dependence of travel time on X2 to examine the potential role of mortality during transport, and discuss situations in which the components of these mechanisms might operate.
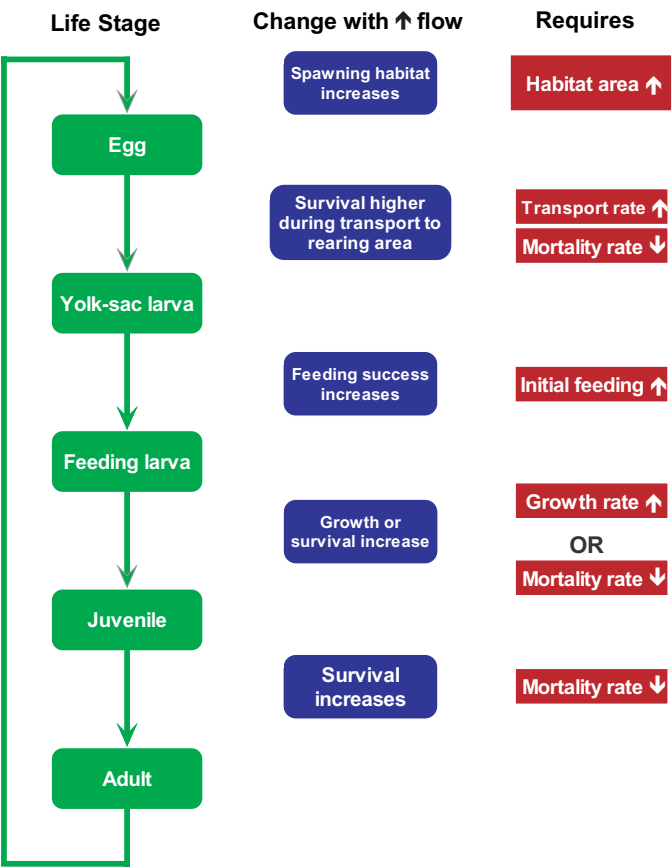

Life Stage	Change with ↑ flow	Requires
Egg	Spawning habitat increases	Habitat area ↑
Yolk-sac larva	Survival higher during transport to rearing area	Transport rate ↑ / Mortality rate ↓
Feeding larva	Feeding success increases	Initial feeding ↑
Juvenile	Growth or survival increase	Growth rate ↑ OR Mortality rate ↓
Adult	Survival increases	Mortality rate ↓

Figure 1 Diagram of classes of fish-X2 mechanisms. The first column indicates life stages of a target species (usually a fish). The second indicates what has to change as flow increases for an X2 mechanism to operate at that life stage. The third column indicates what this mechanism would require in order to operate as hypothesized. This can be useful in deciding how to investigate the mechanisms.

For species that recruit from the rivers (e.g., striped bass and American shad) we estimate travel time by breaking up the distance to be traveled into two segments: from the spawning ground near the mouth of the Feather River to Rio Vista in the Delta, and from Rio Vista to X2. Speed of transport by river flow is approximately a power function of flow (Mount 1995 Eq. 2.1 and 2.4). Sommer et al. (2004) determined mean speed along the Sacramento River from the head of the Yolo Bypass near the mouth of the Feather River to the outlet near Rio Vista. Their data fit a power function:

$$y = 0.006 \, Q_{sac}0.7 \qquad (1)$$

where v is mean velocity (m s-1) and Q_{sac} is Sacramento River flow (m^3 s^{-1}). For convenience a slight neg-

ative curvature (about 1% of the variance in log velocity) is ignored in this function. If this velocity relationship applies over the entire distance D between the spawning area and Hood, the travel time in days (TT) is then:

$$TT = 1.91 \, D \, Qsac^{-0.7} \qquad (2)$$

The relationship between Sacramento River flow and X2 is complex, depending on a variety of other factors including export flow and Yolo Bypass flow. However, for a limited range of X2 values we found that the relationship between Sacramento River flow and X2 was nearly log-linear. We inserted that relationship into equation 2 to obtain travel time as a function of X2:

$$TT = 3 \times 10^{-6} \, D \, X2^{2.1} \qquad (3)$$

We have been collaborating with Matt Nobriga (DWR) to conduct particle tracking model studies using the link-node model DSM2 (Enright et al. 1996). This model was run using constant flows and invariant tides for a variety of inflow and export conditions. A collection of 5000 particles was released at several release points over a 24-hour period. The particles were counted as they passed various points in the estuary. For the purposes of this analysis, we used the release site at Rio Vista and the recovery site at Chipps Island. We then calculated travel time to the LSZ (the region indexed by X2) using the calculated velocity.

Particle tracking runs were made in twenty combinations of inflow, nine values from 340 to 3400 $m^3 \, s^{-1}$, and export flow, five values from 57 to 340 $m^3 \, s^{-1}$, not in all combinations. We ran the model for 92 days, filtered the output to remove tidal fluctuations, and calculated the time at which half of the particles had passed the Chipps Island control point. We then calculated parameters of a model relating the log of travel time to the log of X2 and took antilogs to obtain the relationship between travel time and X2 as a power function. Finally, we calculated travel time to X2 by multiplying the Chipps Island travel time by the ratio of distances traveled. We also determined travel time from Hood to Rio Vista using PTM results to compare with the calculated travel time from the Feather River to Rio Vista.

The resulting travel times show increases with X2, as expected (Figure 2). The travel time determined using the data from Sommer et al. (2004) was very short in all cases, at most ~5 days, which is consistent with their estimate of

hydraulic residence time. The travel time from Hood to Rio Vista based on the particle tracking model was lower than the travel time from the Feather River at high flow (low X2) but increased more rapidly and was much higher at low flow. This may be due to tidal effects which were not included in the calculations of Sommer et al. (2004). In any case, the travel time from Rio Vista to X2 was much longer than either of the other travel times, because of increasing tidal effects and an increasing channel cross-section below Rio Vista. Travel time began to decrease at about X2 greater than ~89 km because the distance from Rio Vista (River Kilometer 100 km) to X2 was shrinking.

Mortality may be higher in the rivers than the LSZ due to higher rates of visual predation in less-turbid water, or due to the physical rigors of transport of eggs and early larvae. There may also be ways that mortality could increase with decreasing flow (Mechanism T2), at least in the rivers. These are discussed below under species-specific considerations, and the mortality rate of striped bass is calculated based on its X2 relationship and these travel time results.

We cannot yet perform an equivalent analysis of transport up the estuary for species that recruit from the ocean. Landward transport by gravitational circulation presumably also scales as a power function of X2 because of the effect of a seaward increase in mean depth resulting in a greater tendency to stratify than expected by theory (Monismith et al. 2002). Particle-tracking studies using the TRIM3D model are being conducted to test this mechanism and these should provide insight. However, if these studies show that speed of transport increases with seaward X2, we will still need to model and then measure the extent to which mortality during transport exceeds baseline mortality in the rearing area.
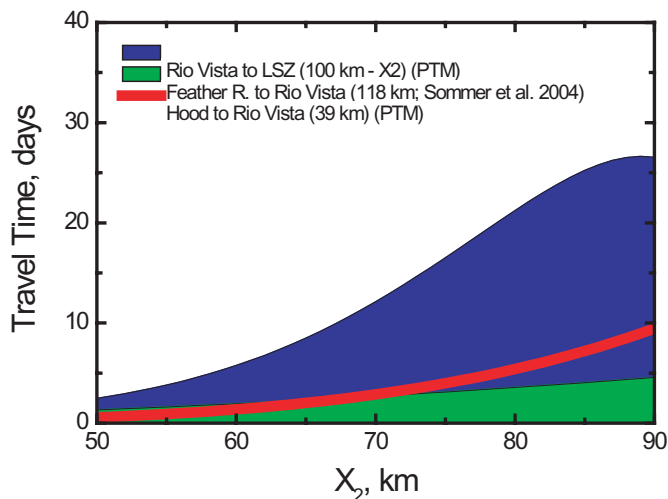
A model fit to the secchi disk data from the DFG zooplankton survey (Figure 3) shows that secchi depths average about 0.2 m in the LSZ at high flow, grading to 0.6 m at low flow and far from the LSZ, either seaward or landward. This is a 3-fold change in water clarity, which could substantially affect predation rates, particularly through light attenuation in deeper water. We have conducted a model analysis of this situation but are still exploring the results and will report on them in a future article.

Figure 2 Travel time based on particle tracking model (PTM) or calculations in Sommer et al. (2004).

Mechanism C: Water clarity is generally lower in the LSZ than in other parts of the estuary, and decreases as flow increases and X2 moves seaward (Kimmerer et al. 1998). Turbidity and low light are associated with reductions in success of visually-foraging fish (Breitburg 1988, Chesney 1989), although studies in real foodwebs can reveal surprising interactions (Cuker 1993).   However, this reduction may be greater for piscivores than planktivores, so that turbid environments are favorable to smaller fish (DeRobertis et al. 2003).

If that is true, then increases in turbidity due to increasing flow should result in better survival of age-0 fish and consequently stronger year-classes when the water is turbid, which generally is associated with high flow. This may reduce predation rates on small planktonic organisms, or it may reduce the risk of attack by visual piscivores such that planktivores can devote more time to foraging and less to predator avoidance, resulting in higher growth rate and better year-class success.

Previous analyses of suspended sediments in various parts of the estuary have revealed the importance of variability at the tidal time scale, and of bathymetric features (Schoellhamer 2001). The link between suspended sediments and secchi depth is not particularly strong, but secchi depth probably gives a better indication of conditions for visual predation than suspended sediments measured at the bottom. Secchi depth should be linearly related to visual detection range for a given predator and prey, and also can be used to calculate light extinction coefficient so that light levels in the water column can be determined.
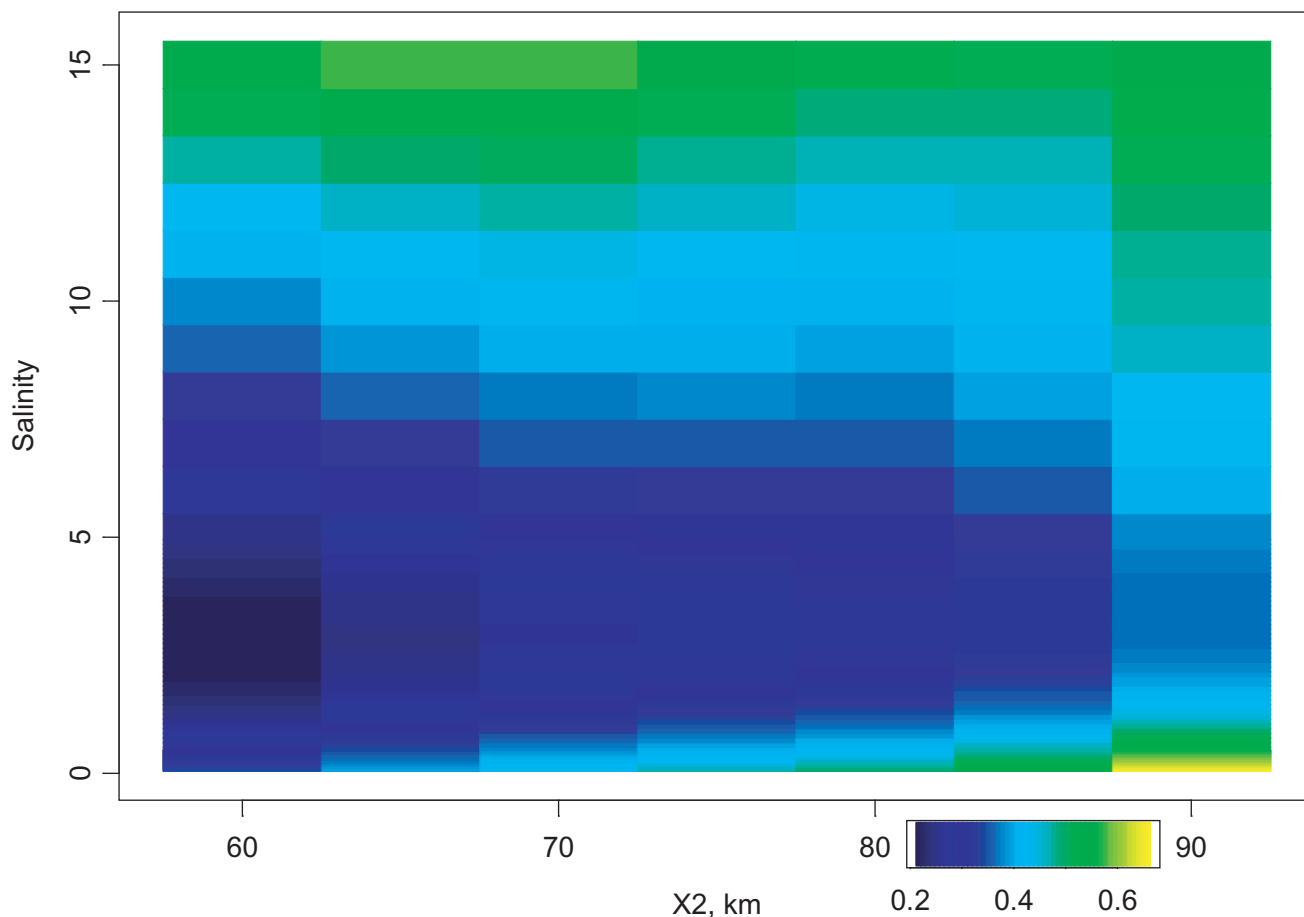
Figure 3 Secchi depth fit to a smoothed curve and interpolated at selected points of salinity and X2. Data from the IEP zooplankton survey.

Mechanism F: Food supply is one of the usual suspects in any analysis of variability in population size. Feeding, particularly early feeding by larval fish, can be an important mechanism for variability in recruitment (de Lafontaine and Leggett 1989). In relation to a potential mechanism for fish-X2 relationships, the principal assumption is that the food supply for a species increases with increasing flow or seaward X2. However, most measures of food for young fish appear not to increase with X2 (Kimmerer 2002a). Furthermore, many of the taxa that provide food for young fish declined during the late 1980s, and in some cases their relationships with X2 changed magnitude or even sign (Kimmerer 2002a). In the Delta, phytoplankton biomass decreases with increasing flow because of decreasing residence time (Jassby et al. 2002). Phytoplankton biomass in the LSZ had no relationship to flow or X2 (Kimmerer 2002a, b).

It could be argued that production could increase even without an increase in biomass. However, this seems unlikely. First, specific growth rate (i.e., per unit biomass) of phytoplankton is related to available light and turbidity (Cole and Cloern 1984, Jassby et al. 2002), and turbidity increases as flow increases (see above). Specific growth rate of zooplankton is controlled by temperature (McLaren 1978) and food concentration (Vidal 1980, Kimmerer and McKinnon 1987), which are unresponsive to X2 if food supply is taken to mean phytoplankton biomass.

Despite the above arguments, it is still possible that food supply for some species increases with increasing flow. The flux of organic matter into the estuary must increase with flow, and may support the microbial food-web. Unfortunately, estimates of microbial activity (e.g., bacteria, ciliates) have not been estimated for more than a brief period. This production is unavailable to most fish

directly and without an increase in zooplankton biomass with flow, there seems to be no way for this mechanism to result in flow-related variability of fish growth. However, we cannot be certain at this point that our measurements are inconsistent with this mechanism, especially because feeding occurs at much smaller scales than we typically average X2 or even measure in the field. Furthermore, this mechanism is so important in other estuaries, and such an obvious candidate ecologically, to be ruled out so early.

Mechanism H: Spatial extent of habitat (area or volume) may vary with X2 or flow. Here habitat may be defined by any suite of variables, but particularly including salinity, temperature, and depth. The clearest example of habitat limitation is the sharp increase in splittail abundance during very high-flow years. This increase is apparently due to the increase in spawning or foraging habitat that becomes available when floodplains are inundated, increasing shallow habitat by orders of magnitude (Sommer et al. 1997, Moyle et al. 2004). At one of our EET meetings Bruce Herbold presented a wide-ranging discussion of possible links between physical habitat and species abundance, but for the moment we focus on relatively straightforward aspect that we can quantify using available information. The more esoteric examples may have to be investigated in the field.

Except where floodplains are inundated at high flows, it is reasonable to expect changes in extent of habitat as defined by salinity to be linearly related to X2. Freshwater habitat must increase, and marine habitat decrease, with a seaward X2. Intermediate salinity ranges may increase or decrease, but the scope for such changes is limited. In any case, a linear change in habitat area or volume is likely to result in a proportional change in abundance of a species that is habitat-limited. It cannot explain an exponential change in species abundance without some concurrent change in habitat quality through one or more other mechanisms, including those discussed here. This means that the change in habitat area per se may be insufficient to produce an exponential increase in abundance.

A more likely scenario may occur with an interaction between bathymetry and salinity, by which the area or volume of habitat within some salinity-depth combination changes sharply with movement of X2. This has been hypothesized to occur when the LSZ moves between the Delta and Suisun Bay, but the density of salinity measurements has been insufficient to establish whether this actually happens or not.

We are investigating this mechanism in collaboration with Ed Gross and others by using the hydrodynamic model TRIM3D to provide the spatial and temporal coverage of salinity distributions with X2. The model will provide tables of volume or area at various combinations of depth and salinity. We will then determine the ranges of these variables that qualify as habitat for various species as percentiles of their distributions and calculate the area and volume within those limits. In addition, we will examine the available distribution data for persistent association of fish with particular locations under different flow conditions.

Species-specific considerations

Some potential mechanisms may depend on specific attributes of species, and it may be possible to isolate the life stage during which a given mechanism acts. Species with similar distributions and life stages may eventually be classified according to particular mechanisms. Here we discuss life history characteristics and explore specific mechanisms that may apply to each species, and attempt to delineate the life stage (Figure 4) and location (Figure 5) of these mechanisms.

Bay Shrimp Female shrimp *Crangon franciscorum* spawn in the ocean and the offspring enter the estuary as juveniles during spring (Hatfield 1985). As with many other estuarine decapod crustaceans, the young shrimp make their way up the estuary and generally are most abundant near the LSZ. Late juvenile and adult shrimp in the LSZ remain near or on the bottom, rising off the bottom on both strong floods and ebbs (Kimmerer et al. 2002). This life history limits potential X2 mechanisms to the juvenile-adult stages, and the locus of the mechanism to the entrance of the estuary, the lower estuary, or the LSZ. Likely mechanisms are: 1) the proportion of the young shrimp population that moves into the estuary increases as X2 moves seaward because of an increasing low-salinity signal inducing migratory behavior, or increasing gravitational circulation; 2) mechanism T (above) for gravitational circulation; or 3) mechanism H. There is no strong evidence for or against any of these, but our hydrodynamic modeling work should be revealing of the possibilities for all three.
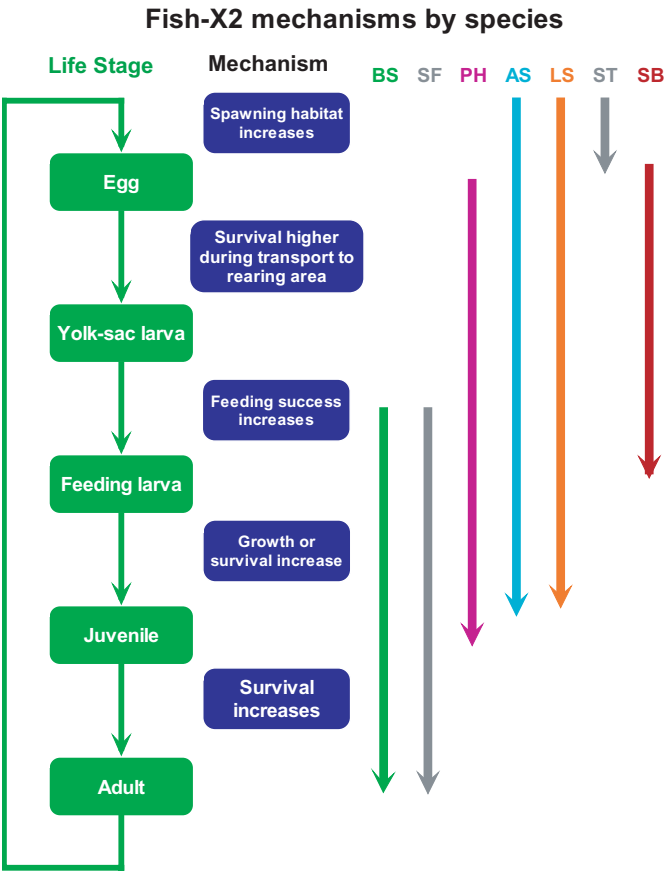
Fish-X2 mechanisms by species

Figure 4 As in Figure 1 showing which species are likely to be affected by flow conditions at which life stage.

Starry flounder has a somewhat similar life history to bay shrimp but the young disperse further up the estuary, and some are even collected at the south Delta fish facilities (Baxter et al.1999). Juvenile flatfish including some flounder are known to undertake tidally-oriented migration to move in a selected direction (Harden Jones 1978). Thus, the mechanisms most likely to be important for starr flounder are similar to those for bay shrimp.

Pacific herring spawn in mid- to late winter on pier pilings, rocks, and vegetation in Central Bay.Maximum hatching success occurs at a salinity of around 16 (Alderdice and Hourston 1985, Cherr and Pillai 1994). Larvae rear mainly in San Pablo Bay and may be food limited some of the time (Gartside 1995). Juvenile herring rear in the estuary at a mean salinity of about 20 psu during the first few months. Thus both hatching success and rearing depend on reduced salinity, and in particular hatching success requires reduced salinity in Central Bay, since there is little suitable habitat in other areas of the estuary. This

is consistent with mechanism H, with one of the key habitat attributes being substrate for spawning.

American shad spawn in freshwater in the Sacramento River basin, essentially overlapping with striped bass. The distribution of juvenile shad is rather different from that of striped bass, though, in that the center of the distribution moves seaward through the estuary as the fish grow (Baxter et al. 1999). This distribution implies that several different mechanisms could be operating; together with the rather weak X2 relationship of this species, this suggests that American shad may not be the best fish to start investigating.
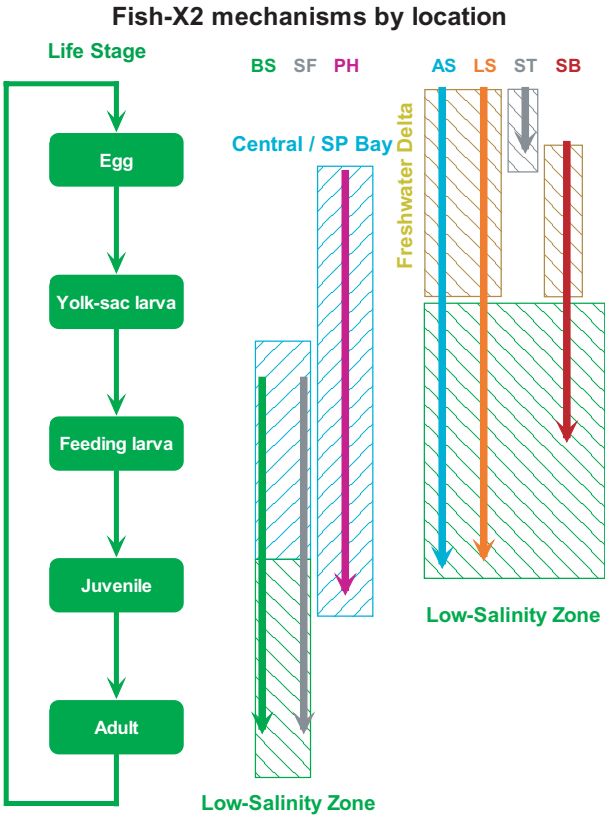
Fish-X2 mechanisms by location

Figure 5 As in Figure 1 showing the locations at which the mechanisms are likely to operate.

Delta smelt are unusual among estuarine-dependent species in having no X2 relationship. This may allow for some productive comparisons with longfin smelt. Delta smelt spawn adhesive eggs most often in the freshwater Delta. Larvae move toward the LSZ during development and juveniles and early adults are concentrated there. Delta smelt are probably the most vulnerable of any of the species discussed here to export pumping, given their distribution during early life. However, their vulnerability depends to some degree on X2, because the population center moves with X2 after the larval stage. This implies either that the effect of entrainment is small, that it occurs before the fish get to the LSZ, or that other factors obscure the entrainment effect.

Longfin smelt have the strongest X2 relationship of any species in the estuary, and also the relationship that changed the most clearly, with a decrease of about 5-fold in abundance for any given X2 value. Longfin smelt spawn adhesive eggs in Suisun Bay and the western Delta during winter. Larvae and juveniles disperse rapidly into San Pablo Bay and further seaward (Baxter et al. 1999), but beginning at the later larval stage these fish begin tidal migrations (Bennett et al. 2002) consistent with attempts to remain within the estuary. These findings seem somewhat contradictory.

Because of the strong relationship with X2, this species should be amenable to investigations into mechanisms. Several possible mechanisms may operate. Transport (T) from spawning areas to rearing areas may be affected by flow, although it is not clear that the requirement for higher mortality during movement, relative to that at the destination, would be met. The spatial extent of habitat (H) has been suggested as a mechanism, but the habitat for this fish is very broad, encompassing essentially the entire estuary below the LSZ (Kimmerer 2002b) and even the coastal ocean (Baxter et al. 1999). The area of habitat within the estuary must therefore shrink when X2 moves seaward. We therefore suspect that some aspect of retention of larvae and early juveniles, related to the strength of gravitational circulation, may be a possibility. Another may be the degree of co-occurrence with food or, alternatively, the availability of food in areas where turbidity increases with increasing flow (Sirois and Dodson 2000).

The contrast between longfin and delta smelt may depend on the location of the fish during rearing. Delta smelt are more closely associated with the LSZ than are longfin smelt. This implies that the mechanism for the relationship of longfin smelt to X2 may depend on conditions over a broader region than the LSZ. (i.e. landward as well)

Splittail increase sharply in abundance when floodplains are inundated, and the area of physical habitat (H) for spawning, rearing, or possibly adult foraging appears to be the key (Sommer et al. 1997, Moyle et al. 2004). This is a good example of an X2 mechanism related to physical habitat, but one that is unrelated to the actual LSZ habitat that X2 indexes.

Striped bass survival from egg to young-of-the-year (YOY) varies strongly with flow or X2, but the slope of the relationship of survival from egg to 6mm larva with X2 is identical to that for egg-YOY survival (Kimmerer et al. 2001). Furthermore, survival from 6mm to YOY is unrelated to flow or X2. This implies that the mechanism for the striped bass X2 relationship occurs during transport down the rivers (T) or at first feeding (F), which occurs at about 5-6 mm. An alternative mechanism arises because some striped bass spawn in the lower San Joaquin River (Turner and Chadwick 1972), from which eggs and larvae are unlikely to be transported to the LSZ under low-flow conditions.

CDFG (1992, p. 13) hypothesized that low flow in the Sacramento River resulted in poor survival of striped bass eggs because they would sink to the bottom, presumably succumbing to low oxygen or physical damage. If true this would be consistent with mechanism T1. If mortality during transport down the rivers does not depend on flow, survival could still be a function of X2 if mortality in the rivers is higher than that in the LSZ. For striped bass, the slope of the X2 regression was -0.027 (Kimmerer 2002a). Although the relationship between travel time and X2 is weakly nonlinear, the average slope is about 0.08 (Figure 2). This implies an added daily mortality rate of 0.33 d-1 in the river (0.027 / 0.08). According to Rose and Cowan (1993), the maximum daily mortality rate of striped bass is on the order of 0.2 d-1. Although the mortality calculated from the X2 relationship and travel time is rather high, these results do not suggest that the mechanism can be ruled out; nevertheless they suggest that a field program may be able to detect the elevation in mortality by this mechanism.

Developing the research plan

We continue to work on the research plan, using the EET as a forum for discussion of mechanisms and possible ways to investigate them. The principles for this analysis and the subsequent research are:

1. Use what we already know. We have a lot of monitoring data, good hydrodynamic models, and considerable information on many of the target species. Some evidence is available to support or refute some mechanisms, or to refine them to the point where key measurements can resolve one or more mechanisms.

2. Design key studies that separate the realm of possibilities. For some species the range of possible mechanisms is broad, but they fall into categories. For example, if otolith studies reveal that the early growth of longfin smelt is independent of X2, then early transport and first feeding could be downgraded as possible contributors to the X2 mechanism, and subsequent focus would be on later stages of the life cycle.

3. The program should be adaptive, not comprehensive. Even where we do not have particularly good information about certain mechanisms, it is not necessary to try to address all of them at once. Instead, we should select subsets of mechanisms for each species or suite of species that are ripe for analysis, and plan efforts to provide needed information in the next few years. If investigations fail to show that selected mechanisms are operating, then we should go to the next most likely mechanism(s).

4. Economies of scale should be considered. For example, a potential mechanism that may involve more than one species should be preferred over one that applies only to one species, all else being equal.

5. Do the easy things first, but begin the long-term studies soon. Some of the mechanisms could be further refined or even tested by data analysis and simulation modeling. This has been started and should start producing answers early in 2005. However, some more labor- and time-intensive studies will most likely be necessary irrespective of the outcome of the analysis and modeling. These studies should be started within a year or two so that results will be available within the time frame of the program.

Our current project will finish in mid-2006, but we anticipate making some recommendations for studies well before that time.

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Vidal, J. 1980. Physioecology of zooplankton. I. Effects of phytoplankton concentration, temperature, and body size on the growth rates of Calanus pacificus and Pseudocalanus sp. Mar. Biol. 56: 111-134.Development and Evaluation of Bootstrapped Confidence Intervals for the IEP Fish Abundance Indices

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Development and Evaluation of Bootstrapped Confidence Intervals of IEP Fish Abundance Indices

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Introduction

The Department of Fish and Game (DFG) conducts numerous trawl-based fish monitoring programs in the San Francisco Estuary (Table 1). These surveys produce a variety of data used in the management and analysis of estuarine fish populations and their responses to freshwater flow and other environmental conditions. Several of these programs develop annual abundance indices for fishes and macrocrustaceans (Baxter et al. 1999). These indices have been centerpieces of numerous publications evaluating natural and human influences on the estuary (e.g., Stevens et al. 1985; Moyle et al. 1992; Jassby et al. 1995; Sommer et al. 1997; Kimmerer et al. 2000, Kimmerer 2002, 2004), but they have been criticized for their lack of a measure of within-year variability. In particular, although analyses of trends are robust, it is unclear to what

extent a change in an abundance index from one year to the next reflects a change in abundance, or simply sampling variability.

Table 1. Description of selected San Francisco Estuary monitoring programs.

Survey	First Year of Survey	Primary purpose(s)
Summer Townet Survey	1959	Abundance indices for 38 mm striped bass and age-0 delta smelt
Fall Midwater Trawl	1967	Abundance indices of age-0 fishes
Bay Study Midwater Trawl	1980	Abundance indices of fishes and macrocrustaceans
Bay Study Otter Trawl	1980	Abundance indices of fishes and macrocrustaceans

Indices are calculated in different ways for each survey, but all involve expanding catch data by an areal or volumetric weighting factor based on estuarine geography or bathymetry (Turner and Chadwick 1972; Armor and Herrgesell 1985; Stevens et al. 1985; Moyle et al. 1992; Dege and Brown 2004). The variance in catches that underlie the indices for a given species can be high because many samples in a survey have zero catch. Furthermore, because many fish species school, occasional very high catches may be common, also resulting in high variance.

Confidence limits for the indices would be useful, but their calculation is complicated by the combination of high inherent variance and the method by which the index values in a given survey are calculated. Furthermore, it is not clear that these computations provide any value above simple statistics such as means of the raw data. In this article we calculate confidence limits for catch per unit effort (CPUE) as mean catch per trawl for several fish species, and show a close correlation between mean catch per trawl and the corresponding index. Bennett (2005) found that CPUE and abundance indices were closely correlated in several sets of monitoring data for delta smelt.

A confidence interval (the interval between upper and lower confidence limits) is defined as that interval with a selected probability (we use 95% out of habit) of containing the mean or other statistic. Confidence limits are considered unbiased if, in repeated samples from the same population, the 95% confidence limits exclude the mean 5% of the time: 2.5% of the time the mean is greater than the upper limit, and 2.5% of the time it is lower than the lower limit. Generally bias arises because the underlying distribution of data is skewed. Although the Central Limit Theorem asserts that the distribution of means of any distribution will approach a normal distribution as sample size increases, for a severely skewed distribution of data the required sample size may be far beyond what a sam-

pling program can afford. Therefore, we should expect asymmetric and possibly biased confidence limits from trawl data.

Several methods are available for calculating confidence limits. These may be parametric, in which the underlying statistical distribution of the data is assumed to take some mathematical form, or non-parametric. Parametric distributions may be difficult to apply to data having a high degree of contagion, although the log-normal and negative binomial distributions have been used successfully (Jahn 1987). Simulations have shown that standard calculations fail to produce unbiased confidence intervals for skewed data even with large sample size, although corrections are available that can alleviate the bias in some cases (Jahn and Smith 1987). When there are many zeros present, as in most of the trawl data from the San Francisco Estuary, it is not clear that any method gives unbiased confidence limits.

The wide availability of desktop computers and modern statistical software have added resampling techniques to the toolboxes available to scientists and resource managers. Resampling techniques repeatedly draw random subsets from the data being analyzed and recalculate mean values based on the subsampled data. A chosen procedure is performed 1,000 times to generate many alternative means. The resulting large sample of simulated mean values can be considered to estimate the probability distribution function of means of samples from the original distribution (below we discuss what that distribution is).

Several resampling techniques have been developed to estimate confidence intervals or other estimators of dispersion for data with non-normal or unknown error distributions (Manly 1997). A widely used method is the bootstrap (Efron and Tibshirani 1993). The bootstrap technique samples a given dataset randomly, but with replacement (i.e., an individual observation may be

picked more than once) to produce an alternative dataset with the same number of observations as the original. This is done repeatedly and the statistic of interest (in our case the mean) is calculated each time.

Although the bootstrap method could be adapted to calculate confidence limits around the indices, we found it simpler and more straightforward to calculate bootstrapped confidence intervals based on CPUE. Thus, the statistic being estimated was the mean catch per tow for selected fishes collected in four surveys (Table 1): the summer townet survey (TNS), fall midwater trawl survey (MWT), and the San Francisco Bay midwater trawl (BMWT) and otter trawl (BOTT) surveys. We show below that the indices and the bootstrapped mean catch per tow are highly correlated and therefore can be considered estimates of the same variable. Whether that variable is actually population abundance is another matter. Bootstrap methods are not a way of avoiding the problems inherent in sampling from patchy distributions, and to some extent they may exacerbate these problems. This analysis cannot address the adequacy of these sampling programs for characterizing the actual abundance of fish.

Methods

Raw catch data were obtained from the four surveys for several estuarine fish species, some of which were sampled in more than one of these programs. For each of these species we also obtained abundance indices calculated by the Department of Fish and Game (Turner and Chadwick 1972; Stevens et al. 1985; Moyle et al. 1992; Baxter et al. 1999).

Data from the TNS were reduced to include only stations that had been sampled through most of the time series. Station 323 in San Pablo Bay was sampled starting in 1967 and continues to be sampled; all stations sampled during at least that time period were included. Data for striped bass were taken only from the last two surveys of each year to correspond to the index (Turner and Chadwick 1972). Data for delta smelt were taken from the first two surveys for the same reason (Moyle et al. 1992). No other fish were abundant enough to be included in this analysis.

Data from the FMWT survey were used in their entirety. Data from the Bay Study included only stations (called Series 1) sampled throughout the history of the

program, which have also been used for calculating indices (Orsi 1999). The time period for calculating catch per tow was the same as that for calculating the indices, most often July - October. Calculations were made separately for the midwater and otter trawls from the Bay Study.

For each year in the data record, we took all of the samples meeting the criteria to be included in the analysis, as described above. We then calculated bootstrap means and bias-corrected 95% confidence limits (Efron and Tibshirani 1993) using the function "bootstrap" in S-Plus version 6.2 for Windows. In this procedure, samples of the same size as the original sample are taken from the available data with replacement, a mean is calculated, and the process is repeated 1000 times. The mean of these 1000 values is the overall bootstrap mean. Bias-corrected confidence intervals compensate for skewness in the underlying distribution by calculating percentiles that are adjusted up or down depending on the degree of skewness. Without skewness, these percentiles are 2.5% and 97.5% for the 95% confidence intervals.

Linear regressions with zero intercept were then calculated between the bootstrap means and the indices for each species in each survey. We prepared graphs of mean catch per trawl over time with confidence limits, with an overlay of the abundance index scaled according to the regression slopes. The FMWT data were used to generate boxplots of frequency of occurrence of selected species at the stations sampled to evaluate qualitatively the influence of fish distribution on catch variability.

We tested the potential effects of skewness on bootstrapped confidence limits by a simple simulation. Two examples of data from the Bay Study otter trawl data set were used, one for a common species (longfin smelt) and another for a relatively uncommon species (age-0 starry flounder). We took data for a single year (1999), removed the zeros, and calculated the mean of the remaining data after log transformation. We then made a simulated data set with similar characteristics, i.e., the same proportion of zeros, and the same mean of the log-transformed data with zeros removed, but with 1,000,000 observations. This was then the "population" from which we sampled. We took 5000 samples from this population with an N of 60 (close to the low end of the range of sample sizes in the actual data) or 200 (the maximum for the Bay Study data) and for each sample we calculated bootstrapped 95% confidence limits as above. We then tabulated the frequency

with which the actual population mean exceeded the confidence interval in either direction.

Results

All of the grand mean CPUEs based on bootstrapping were significantly correlated with the official abundance indices (Table 2; Figures 1-4). Most correlation coefficients were ≥ 0.95; only two, longfin smelt in the BMWT and delta smelt in the TNS were < 0.90 (Table 2). In all surveys the confidence limits were narrow enough that individual years could be clearly distinguished (Figures 1-4)

Table 2 Correlations between abundance index and mean catch per trawl from the bootstrap analysis for selected species in the four sampling programs.

Survey and Species	Correlation coefficient
Summer Townet Survey	
Delta smelt	0.83
Striped bass	0.90
Fall Midwater Trawl	
American shad	0.99
Threadfin shad	0.95
Delta smelt	0.96
Longfin smelt	0.98
Splittail	0.97
Age-0 striped bass	0.96
Bay Study Midwater Trawl	
American shad	0.99
Delta smelt	0.95
Longfin smelt	0.83
Striped bass	0.99
Bay Study Otter Trawl	
Longfin smelt	0.96
Striped bass	0.97
Age-0 starry flounder	0.98
Age-1 starry flounder	0.97
Pacific staghorn sculpin	0.94
Yellowfin goby	0.99

Each survey had species that could be characterized as comparatively "high" versus "low" variability species (Figures 1-4). Using the FMWT as an example (Figure 2), American shad, longfin smelt, and striped bass had low catch variability, as shown by the confidence intervals, compared to threadfin shad, delta smelt, and splittail. Relative abundance did not explain this pattern, as both the high and low variability groups contained species with high relative abundance. Rather, the evenness of a species' distribution appeared to have a greater effect on catch variability since American shad, longfin smelt, and striped bass were on average collected at a greater number of sites than threadfin shad, delta smelt, and splittail (Figure 5).

The simulations showed that lower confidence limits were conservative but that there was some bias, and particularly that the upper confidence limits excluded the mean more often than expected (Table 3). This suggests that the actual upper confidence limits are at the 5-10% level rather than the 2.5% level as intended. Attempting to correct for this bias would put too much emphasis on the numeric value; rather, readers should be aware of this bias and take it into account when inspecting the figures.

Table 3 Simulations of selected sampling distributions from selected data on longfin smelt and age-0 starry flounder from the Bay Study otter trawl. The number in sample is the size of the sample from a simulated distribution of CPUE with N=1,000,000 and distributions similar to those of the actual data. Other columns show the grand mean of the distribution from which samples were taken, percent of the 1000 or 5000 bootstrapped samples whose calculated confidence limits did not include the grand mean; and the percent of the original data (and the simulated distribution) that had zero catch.

| | | | Percent outside confidence limits | | | |
| | | | Percent | | | |
Species	Number in Sample	Number of bootstrap samples	2.5% limit	97.5% limit	Grand Mean	Zeros
Longfin smelt	60	5000	2.1	9.4	8.5	57%
Longfin smelt	200	1000	2.0	5.6	7.3	57%
Starry flounder	60	5000	2.0	6.5	0.5	83%
Starry flounder	200	1000	2.0	5.3	0.5	83%

Discussion

Two general conclusions can be drawn from this analysis. The first is that CPUE and abundance indices provide the same information about fish populations in the estuary. The second is that confidence limits around estimates of mean CPUE are not excessive, even considering the effect of skewness discussed above, and differences in abundance among years can be readily distinguished.

The high correlation coefficients between the official IEP abundance indices and the underlying CPUE suggests that the steps used to convert CPUE into indices, including areal or volumetric expansion factors and summation across surveys, are unnecessary. We suggest that the grand mean catch per tow is a superior index of relative abundance with several advantages over the indices. First, catch per tow is directly available from the raw data. Thus, the relationship between the data collected and the index produced would be as clear as possible. Second, confidence intervals can be associated with the indices as shown above. Third, comparisons among surveys would be more straightforward, and might indicate important attributes of species' patterns of distribution or abundance. Fourth, to the extent that analyses require abundance estimates (e.g., in terms of biomass) rather than indices, catch per tow provides the basis for these estimates.

The confidence limits around catch per trawl estimates were not excessive (Figures 1-4), no doubt because of the intense and consistent sampling effort. We commend the original designers of these programs and the people who maintain them for this level of effort, which makes these data so valuable.
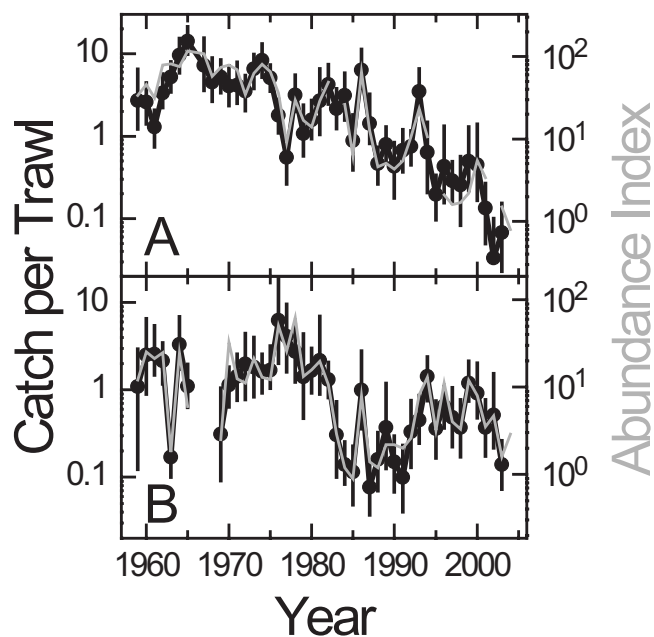

Figure 1 Summer townet survey. Mean catch per trawl with bootstrapped confidence limits (left axis, error bars and thick line), and abundance indices (thin line, right axis). Only two surveys (of 2-6, median 4) were included in each estimate: A = striped bass, B = Delta smelt.
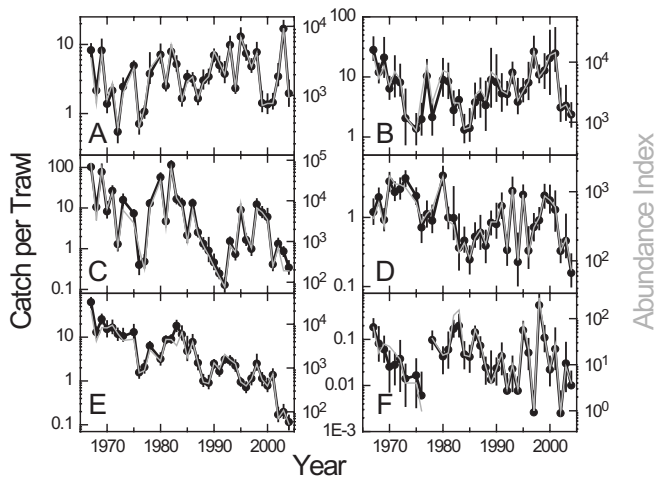
Figure 2 Fall midwater trawl. Symbols as in Figure 1: A = American shad, B = Threadfin shad, C = Longfin smelt, D = Delta smelt, E = Striped bass, F = Splittail.
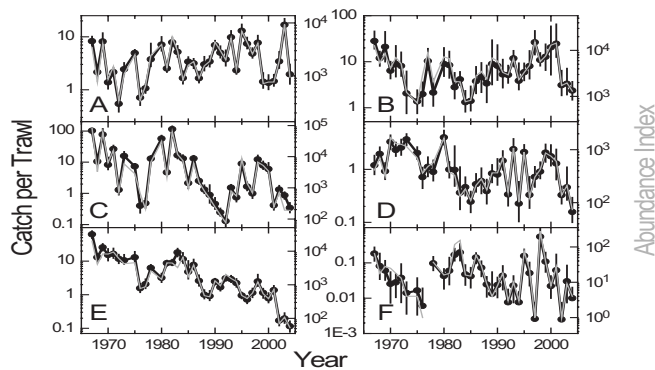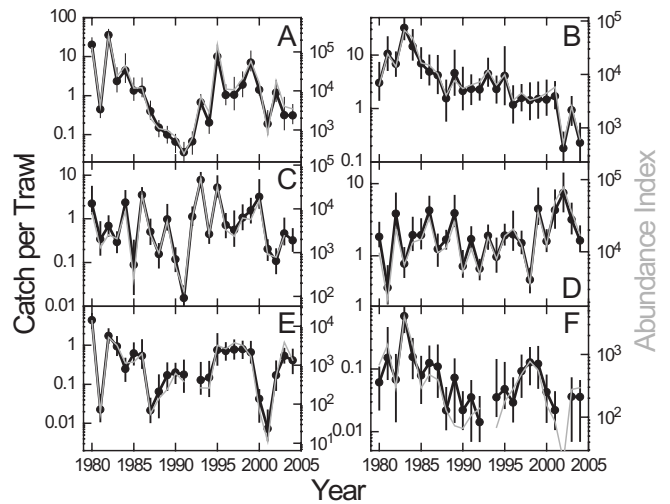

Figure 4 Bay Study otter trawl. Symbols as in Figure 1: A = Longfin smelt, B = Striped bass, C = Yellowfin goby, D = Pacific staghorn sculpin, E = Age-0 starry flounder, F = Age-1 starry flounder.
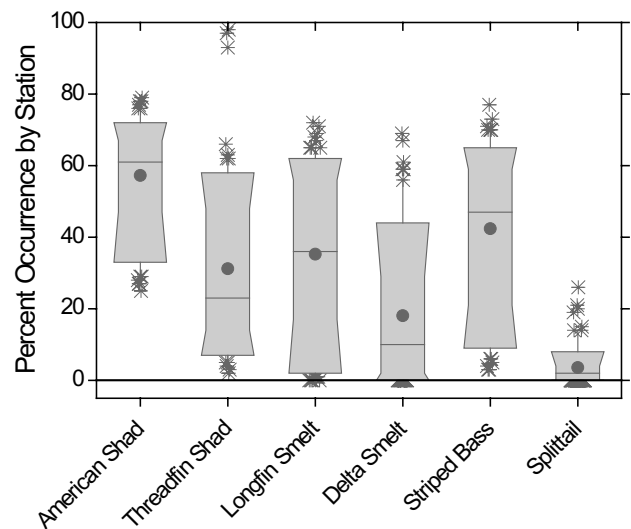

Figure 3 Bay Study midwater trawl. Symbols as in Figure 1: A = American shad, B = Striped bass, C = Longfin smelt, D = Delta smelt.

Figure 5 Fall midwater trawl. Presence / absence of species in Figure 2, presented as percentage of samples at each station in which at least 50 samples were taken over most of the period of the survey. Boxplots give mean (filled circle), median (horizontal bar), and 10, 25, 75, and 90 percentiles of the data (box corners and notches). Whiskers indicate 5th and 95th percentiles, and outliers are shown as asterisks.

The rather narrow confidence limits around the CPUE values allow conclusions to be drawn about differences among years, even in the presence of the bias shown in Table 3. For example, recent low abundance values have raised the question whether the indices are sufficiently reliable to claim a change in abundance based on only one

or a few years of data, as opposed to a long-term trend. Taking striped bass in the MWT survey as an example, the mean catch per trawl during each of the last 3 years was clearly lower than in any of the previous years, taking confidence limits into account. The picture from the other surveys is somewhat different, but still the data show that the last 3 years have not been good for striped bass. The same cannot be said for other species individually; for example, although the MWT index (and CPUE) for delta smelt has been low, none of the values is without historical precedent and all of the recent values have confidence intervals that overlap those of previous years. Likewise, the CPUE of threadfin shad in recent years is well within the range of values seen previously. Note that we did not attempt to evaluate CPUE across species, so we have not determined whether abundance of pelagic fishes as a group has been at an unprecedented low over the last few years.

Low variability in CPUE for some species arises partly because of high abundance but perhaps more so from a wide geographic and temporal distribution, which results in relatively few zeros in the data (compare Figures 5 and 2). This provides insight into how best to reduce catch variability for more narrowly distributed species. We made no attempt to exclude stations from which a species was never collected when estimating confidence intervals. If a species was never collected from stations in a certain region, or perhaps with a certain set of characteristics such as depth or salinity, removing those stations before calculating mean catch per tow would decrease overall variability. More sophisticated approaches also may be possible. For instance, allowing the stations included in the index to vary from year to year based on an empirically determined salinity range may remove additional zero catch data, further reducing variability.

In any analysis it is helpful to think about the scope of inference for the conclusions drawn from the data. In the case of confidence limits around mean CPUE, strictly speaking inferences are in relation to the mean of catches of fish from the same suite of stations during the same months with the same gear. A stratified random design with suitable adjustments for capture efficiency would be required for estimating abundance. We believe that for highly mobile fish species the existing design may be adequate for estimating abundance, but that should be tested, and it may not be true for species that have some orientation to bathymetric features. Delta smelt aggregate in the lower Sacramento River during low-flow periods (Sweet-

nam 1999; Bennett submitted) which may preclude estimating their abundance without an alteration of the sampling design.

Our recommendations for IEP, then, are:

1. Replace abundance indices with CPUE and provide confidence limits with all estimates.

2. Conduct a study to assess the validity of the fundamental assumption underlying fish sampling, i.e., for which species is the sampling effectively stratified random with respect to the distributions of fish.

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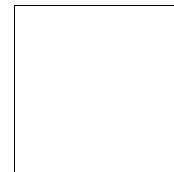
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